Monetary and Social Valuation: State of the Art

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1.1 Why valuation?

Ecosystems provide important commodities and environmental benefits to society. As such, the management of ecosystems is an economic, social and political issue encompassing all sectors of an economy. It involves trade-offs between competing uses and users, as well as between additional economic growth, ecosystem protection, and further natural resource depletion and degradation. Any particular use of ecosystems has its opportunity costs, consisting of the foregone benefits from possible alternative uses of the resource. Decision-makers are faced with balancing these varied resource uses, for example, between fresh water demands from agricultural irrigation for food production on the one hand, and the desire to protect rivers for fish and wildlife habitat by maintaining environmental flow levels on the other. Striking a balance in the trade-offs between economic growth and ecosystem resource use possibly leading to their degradation and depletion is crucial for the sustainable management of our natural resources. Economic valuation contributes to an improved natural resource allocation by informing decision-makers on the full social costs of ecosystem exploitation and the full social benefits of the goods and services that healthy ecosystems provide.

The general idea behind putting a monetary value on ecosystem goods and services is to allow for more informed and eventually more efficient trade-offs between all of societies' scarce resources, i.e., including ecosystem resources. With that in mind, the most common justifications for economic valuation of ecosystem services are advocacy (advocate the economic importance of the environment), influence decision making and policies, calculate damages for liability compensation, inform policies aimed at internalizing negative and positive externalities. With respect to the latter, valuation of ecosystem services can be used to set taxes, fees or charges for the use of those services. Setting taxes, fees or charges have a double role in terms of environmental management. They help to control the extent to which environmental resources are exploited (i.e., the higher the costs of using an ecosystem, the lower its use) and simultaneously may generate revenues that can be used to pay for the management, protection and restoration of the ecosystem. Valuation results can be used to set taxes or charges at the most desirable resource use level based on scientific understanding of for example the resource's natural regenerative or

[1] The majority of this chapter is taken from: Van Beukering PJH, Brouwer R, Koetse MJ, 2015, Economic Values of Ecosystem Services, in: JA Bouma, PJH van Beukering (eds.), *Ecosystem Services: From Concept to Practice*, Cambridge University Press, Cambridge, pp. 89–107.

carrying capacity. Government revenues obtained by the taxation can be redistributed to the people, e.g., by income taxes.

1.2 Typology of values

Ecosystems are natural assets that create flows of goods and services over time. The key to their valuation is to establish the functions that they provide and to link these functions to their societal values. If that link can be established, then the value of a change in the functions provided can be derived from the change in the value of the stream of benefits. Given the multi-faceted nature of benefits associated with ecosystems, there is a need for a useable typology of the associated values. We therefore need to consider how and to what extent the concept of economic value captures the variety of ecosystem values.

Although a number of classification systems exist to describe the different types of values associated with the goods and services provided by ecosystems, economists have generally settled for a taxonomy based on the concept of Total Economic Value (TEV), which is equal to the sum of the components presented in Figure 1.

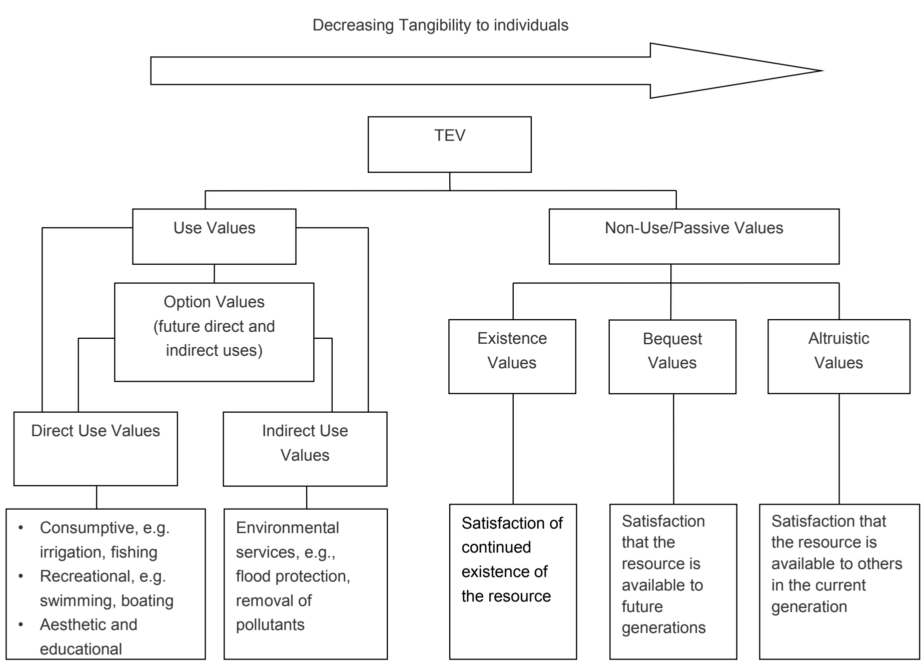

Figure 1. Components of the Total Economic Value of ecosystem services (source: Based on Brouwer et al., 2009)[2]

[2] Option values include potential values of unplanned and uncertain uses.

The key distinction in Figure 1 is between use values and non-use values. Use values are values directly related to the use of ecosystem goods and services by humans. Three types of use values can be distinguished:

- *Direct use values* arise from direct interaction with ecosystems. They may be extractive, such as the felling of wood for the timber industry and the harvesting of fish, and they may be non-extractive, such as recreational swimming and the aesthetic value of a natural view.
- *Indirect use values* are associated with services that are provided by resources but that do not entail direct use. They are derived or enjoyed indirectly, for example, from flood protection provided by mangroves and coral reefs or the removal of pollutants and greenhouse gas emissions by wetlands and forests.
- *Option values* are values of potential but as yet undiscovered use in the future. They may include values held by the current generation as well as by the next generation, and are related to uncertainty about future demands.

Non-use values reflect values that are not in any way related to the current or future use of an ecosystem good or service, but are derived from the knowledge that an ecosystem exists and is maintained. They are not associated with tangible benefits that can be derived from it, although users of a natural resource or ecosystem service may also hold non-use motivations for its preservation. Therefore individuals may have little or no use for a given ecological asset but nevertheless experience a ‘loss’ if it were to disappear. Examples include the value people attach to the preservation of charismatic species such as the blue whale or important wildlife habitats such as the tropical forest on Borneo or Sumatra in which the orangutan lives, even though they never saw these species or visited the places where they live. Non-use values are often also linked to ethical concerns (e.g., animals also have rights) and altruistic preferences (e.g., ecosystems and their services should be there for others to enjoy too or for future generations). Although it can be argued that these values and preferences ultimately stem from self-interest, at least their existence shows that people’s utility functions may include animal other people’s well-being as well. The boundaries of non-use values are not always clear-cut. Although preferences may reveal important parts of non-use values, some human motivations for the notion that environmental assets should be conserved ‘in their own right’ are arguably outside the scope of conventional economic thought (e.g., those related to ethics and morality). In practice, an important issue is also whether it is possible for individuals to assign meaningful non-use values to environmental assets, or even to express meaningful preferences for them. Non-use values can be divided into three types of value, which can be overlapping:

- *Existence value* is the satisfaction derived from the knowledge that ecosystems and biodiversity exist and will continue to exist, regardless of whether or not they have any use value. Motivations here could vary, but might include having concern for the asset itself or a motive whereby the “valuer” feels some responsibility for the asset.

- *Bequest value* is the satisfaction derived from ensuring that ecosystems and biodiversity will be passed on to future generations so that they will have the opportunity to enjoy it.
- *Altruistic value* is the satisfaction derived from ensuring that ecosystems and biodiversity are available to other people in the current generation.

Economic values can now be combined with the ecosystem services approach to provide a comprehensive economic assessment framework for ecosystem valuation. It is important to note that what is being valued is not the ecosystem per se (i.e., its intrinsic value), but rather the goods and services provided by ecosystems that are beneficial to human beings. Valuation therefore inherently reflects an anthropocentric approach, that is, the values that humans attach to the environment and the services provided. The subdivision of economic values discussed above represents the standard categorisation of ecosystem related values by environmental economists. It is interesting to relate these values to the categorisation of ecosystem services as defined in the Millennium Ecosystem Assessment (2005), which makes a distinction between provisioning, regulating, supporting and cultural services. These relationships are presented in Figure 2.

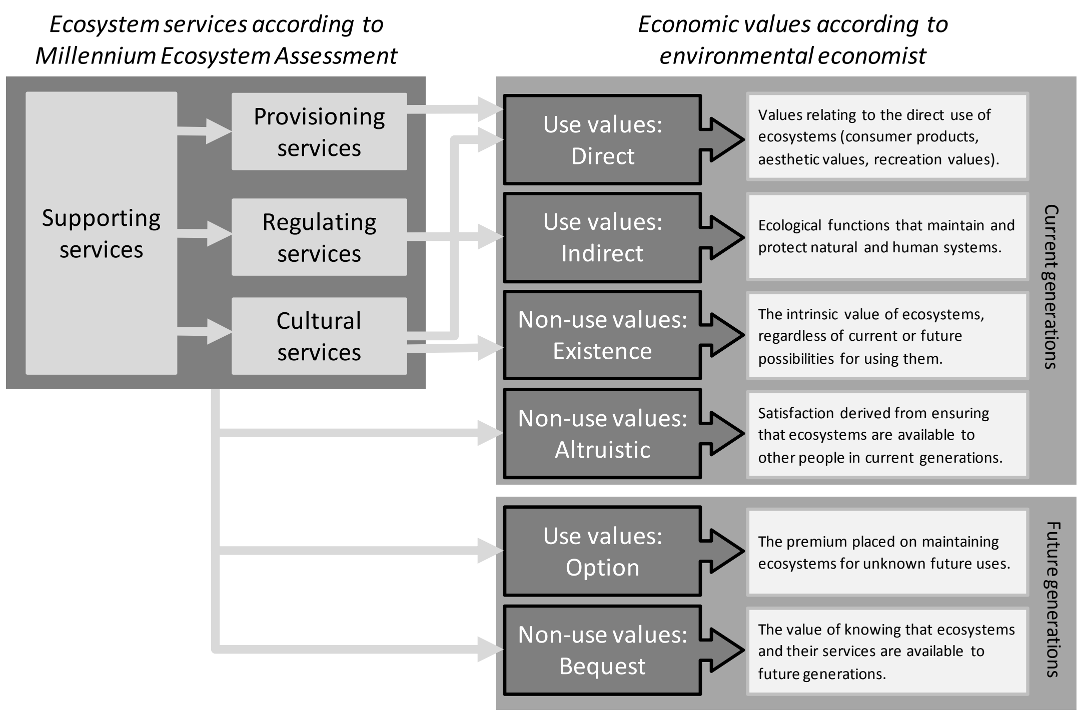

Figure 2. Parallel between ES categorisation of the Millennium Ecosystem Assessment (2005) and the standard value categorisation used by environmental economists (source: Van Beukering et al., 2015)[3]

[3] In this figure we chose to deviate slightly from other definitions on option and bequest values (see, e.g., Kumar, 2010, Chapter 5; MEA, 2003, Chapter 6). The main difference is that we include option values as values relating to *unknown ecosystem services* to future generations, and bequest value as the value of knowing that *known ecosystem services* are available to future generations.

The figure shows that provisioning services and cultural services provide direct use values, that regulating services mainly provide indirect use values, and that existence values are mainly related to cultural services. The figure also shows that all four service categories, or rather ecosystems as a whole, may have altruistic, bequest and option values. The aggregation of all service values provided by a given ecosystem yields the TEV of that ecosystem. In conclusion, a typology of values based on the TEV concept is considered a useful way to represent and provide insight into the multi-faceted nature of the benefits associated with ecosystems, despite some grey areas around the precise demarcation of use and non-use value categories.

1.3 Calculating welfare effects

In calculating the welfare effects from, for example, a policy measure that causes changes in the provision level of ecosystem services, valuation of these changes is not sufficient. For deriving changes in total welfare it is essential that the following effects and quantities are known:

- The biophysical change in quantity and/or quality of the ecosystem service;
- The practical implications of this physical change on different stakeholder groups and their welfare in terms of changes in the relevant use and non-use values involved;
- The welfare change in terms of either individual willingness to pay to prevent the change in the ecosystem service, or individual willingness to accept compensation in case the change would actually occur, depending on the distribution of the property rights across the different stakeholder groups involved;
- The number of stakeholders affected by the change (e.g., visitors to a natural area, the public at large (households), private companies, farmers, etc.) for aggregation purposes.

Ultimately, the welfare effect per individual (e.g., visitor, household, farmer) is obtained by multiplying the change in the ecosystem service by individual WTP or WTA. The total welfare effect is calculated by multiplying the individual welfare effect by the number of individuals affected by the change. This assumes a constant unit value per individual stakeholder. However, in practice, this is typically not the case and requires careful attention and thought when aiming to use valuation results in economic cost-benefit analysis underpinning policy and decision-making.

In many cases the changes in quality and/or quantity of the ecosystem service are known, or can be estimated using existing models. For example, the WTP per household for accepting or preventing a change in an ecosystem service can be obtained by applying one of the available valuation methods (see chapter 2). The number of households affected by a change in the service maybe be difficult to determine. Brouwer et al. (2009) state that determining the relevant market size is particularly difficult , especially in the case of non-use values, and when complications arise due to the distinction between user and non-user vales. In practice, it is often the market size used in studies that receives most criticism when using non-market valuation study results to inform policy and decision-making. Where total welfare estimates are to some extent sensitive to the average WTP values obtained from a valuation study, they are highly sensitive to the population of beneficiaries across which these WTP values are aggregated.

Although there are many relevant issues that should be accounted for in calculating welfare effects, we will address four of the most important ones. First, it is important to account for the spatial distribution of the provision of the ecosystem service (supply) and the spatial distribution of the beneficiaries of (a change in) ecosystem service (demand). A common approach to define the relevant market size is through the identification and estimation of a spatially sensitive valuation function. Such an approach is able to adjust for the notion that underlying values for changes in ecosystem services are likely to decay with increasing distance from the ecosystem, which is relevant for use values but may also hold for non-use values. Aggregation procedures need to be able to recognise and address this problem in order to produce reliable valuation outcomes. The use of a spatially sensitive valuation function explicitly incorporates so-called distance decay relationships into defining the limits of the economic market size. It also allows for variability in the socioeconomic characteristics of the encompassed population within the aggregation process. The economic market is defined by the area within which there are positive values for the ecosystem service involved. The maximum limits of this area can be found, given a certain distance decay relationship, by predicting the point at which WTP values decline to zero.

A second issue is preventing double counting. The ecosystem function approach is used to identify ecosystem goods and services. However, if the value each of these goods and services is identified separately, and then attributed to underlying functions, there is the likelihood that benefits will be double counted. Benefits might therefore have to be explicitly allocated between functions. Some functions might also be incompatible, implying that combining their values would overestimate the feasible benefits to be derived from the ecosystem. In practice, the ability to use ecosystem resources repeatedly or simultaneously for different uses means that competition and complementarity are important considerations in valuation. This means that total economic valuation is undertaken only when necessary. Value assessments are more commonly based on partial valuation, based on a sectoral approach (focusing on a specific stakeholder group) or based on changes in a specific set of goods and services.

A third issue is accounting for possible substitution between ecosystem services. Ecosystems provide different goods and services, some of which may be more or less replaceable through the use of other ecosystems located nearby or further away. Substitution therefore occurs when a negative change in an ecosystem service can be (partially) offset by using the same service provided by another ecosystem, thereby decreasing the negative welfare effect related to the change. For example, recreation in a certain forest may be substituted by recreation in another nearby forest with minimal loss in overall welfare. Research suggests that the value added of incremental ecosystem services provision decreases rapidly once a certain level of ecosystem service provision has been reached (Bateman et al., 2002).

A fourth and final issue is accounting for both positive and negative spatial spillovers. Quality improvements in ecosystem services may affect service quality in other areas when ecosystems are interdependent. This is especially relevant where water ecosystems are concerned, due to the spatial (upstream-downstream) interdependencies of the water system. Not accounting for the benefits or costs that occur in ecosystems or regions outside the original study area may lead to severe over- or underestimation of welfare effects.

1.4 Aims and outline of the deliverable

As discussed in the previous sections, many pieces of information and data are needed for calculating the total welfare effects of changes in ecosystem services. Moreover, many issues need to be addressed to prevent over- or underestimation. In this deliverable the emphasis is on economic valuation, and it aims to achieve three goals: (1) to give an overview of monetary (and non-monetary) valuation methods, (2) to provide insight into recent developments in monetary valuation, and (3) to provide an overview of developments in natural capital and ecosystem service accounting.

The outline of the deliverable is as follows. Chapter 2 presents a review and assessment of economic valuation methods. In order to recognise that are many policy contexts and decision support systems and tools, and to take account of the notion that economic valuation may not be possible or suitable in all situations, Chapter 3 discusses innovative non-monetary methods for measuring socio-cultural values. Chapter 4 contains several contributions on developments in monetary valuation and valuation methods. Chapter 5 discusses the structure and characteristics of a recently developed integrated ecosystem service assessment model. It furthermore presents results of an application of this model in the Scottish national exemplar. Chapter 6 provides an overview of developments in natural capital and ecosystem service accounting. Finally, Chapter 7 discusses relevant further research on the topics addressed in this deliverable, with a focus on research planned within the OPERAs project.

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2 Economic valuation methods[4]

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2.1 Introduction

Various economic valuation methods exist and have been applied to estimate the values of various ecosystem services. The methods and techniques reflect the extent to which the services provided by ecosystems touch on the welfare of society, either as direct determinants of individuals' well-being (e.g., as consumer goods) or via production processes (e.g., as intermediate goods). The aim here is to provide an overview of available valuation methods, to discuss their advantages and disadvantages, and to provide guidance on when to use which method. In doing so we do not aim to be comprehensive; extensive details of the underlying theory and on the actual practice of applying the valuation methods, are provided in general texts, including Bateman et al. (2002), Kanninen (2007) and Bockstael and McConnell (2007).

An important distinction is between market-based and non-market based valuation methods. Market based valuation means that existing market behaviour and market transactions are used as the basis for the valuation exercise. Economic values are derived from actual market prices for ecosystem services, both when they are used as inputs in production processes (production values) and when they provide direct outputs (consumption values). By observing how much of an ecosystem service is bought and sold at different prices, it is possible to infer directly how people value that good. Examples of market based methods are the use of direct market prices, net factor income and production function methods. Cost-based methods, such as replacement costs, defensive expenditures and avoided damage costs, also use market prices. However, these methods are fundamentally flawed as measures of value, and we will discuss them separately.

Unfortunately, direct markets for many ecosystem goods and services do not exist, and direct market prices are missing. In these cases the changes in the supply and/or quality of ecosystem goods and services are often valued through indirect market valuation methods, also referred to as revealed preference methods. Revealed preference (RP) methods are based on actual consumer or producer behaviour and identify the ways in which a non-marketed good influences actual markets for some other good. Preferences and values are 'revealed' in complementary or

[4] A slightly different version of this chapter has been published as Koetse MJ, Brouwer R, Van Beukering PJH, 2015, Economic Valuation Methods for Ecosystem Services, in: JA Bouma, PJH van Beukering (eds.), *Ecosystem Services: From Concept to Practice*, Cambridge University Press, Cambridge, pp. 108–131.

surrogate markets. These methods use data on actual choices made by individuals or firms in related markets. The most important revealed preference methods are the hedonic pricing method and the travel cost method.

Stated preference (SP) methods use surveys to ask people to state their preferences for hypothetical changes in the provision of environmental services. This information on preferences is then used to estimate the associated values that people attach to the environmental services under study. The most important stated preference methods are the contingent valuation method and choice modelling or conjoint analysis.

Two alternative methods, which do not belong to one of the previous method categories per se, are meta-analysis and value transfer. These methods are strictly speaking not valuation methods in themselves because they make use of insights from previous studies. However, since they are often used to derive ecosystem service values, we discuss them anyways. The methods that are generally used for the valuation of ecosystem services and that are discussed in this chapter are summarised in Figure 3.

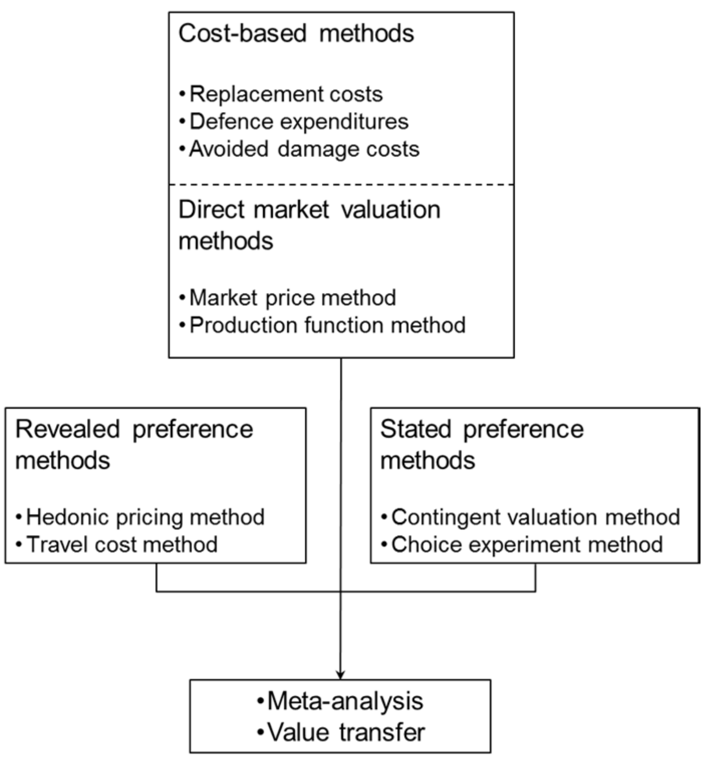

Figure 3. Typology of economic valuation methods (source: based on Figure 6.1 in Koetse et al., 2015)

2.2 Direct market valuation methods

The values of some ecosystem services can be measured using direct market valuation methods. Three basic methods can be distinguished: (1) market price method, (2) production function method, and (3) cost- based methods. Below we discuss these methods in more detail.

2.2.1 Market price method

For some ecosystem products commercial markets exist, and their economic values can be derived by looking at actual market transactions. For some ecosystems services it is not necessary to use complicated valuation methods because they are traded on markets and their economic values can be derived by looking at actual market transactions. Good examples are products such as timber, fuel wood, fish, and other foods. In a competitive market without market failures, market prices accurately reflect the marginal value of an ecosystem service (i.e., the value of a small change in the provision of that service) and resource rents are maximised. As with any other market good, total economic value is obtained by estimating and adding consumer and producer surplus.

An **advantage** of this method is that it is relatively easy to apply, as it makes use of generally available information on prices, quantities and costs, and only requires simple modelling and few assumptions. Also the method uses data on *actual* consumer behaviour and preferences, in contrast to non-market methods that use data on *stated* consumer behaviour and preferences.

A **disadvantage**, although not of the method itself but rather of its applicability, is that many environmental services are not traded directly in markets. Also, if markets for environmental services do exist but are highly distorted, the available price information will not accurately reflect social and economic values. Important sources of such market distortion are taxes and subsidies, non-competitive markets, imperfect information, and government controlled prices (Krugman et al., 2010). Under these circumstances shadow prices can be used instead, although admittedly may be based on other methods than the market price method. Finally, the method cannot be easily used to measure the value of larger scale changes that are likely to affect the supply and demand for a good or service.

Example Box 1: Economic importance of the Caroni swamp in Trinidad and Tobago

The Caroni swamp consists of tidal lagoons, marsh land, and mangrove forests. It provides a number of important ecological and economic functions, including habitat and nursery support to fisheries, forestry products, and recreational opportunities such as bird watching and sport fishing. The economic value of the timber and fuel wood taken from the mangrove forest has been estimated by the direct market valuation method. The results show that the value of these specific services amounts to approximately US$ 4 per hectare of mangrove per year.

Source: Ramdial (1975)

2.2.2 Production function method

Some ecosystem services are used as inputs in production processes, and their values can be obtained by measuring their contribution to the economic value (consumer + producer surplus) of the final good through production functions. A production function estimates the functional relationship between inputs and outputs in production. For example, the number of diving trips to a coral reef can be considered a function of the quantity and quality of the coral reef itself, and the labour and equipment needed to provide the diving service. Also the production of fruits and nuts from a forest may be described as a function of hours spent harvesting (labour) and the area and quality of the forest. To apply this method data must be collected regarding how changes in the quantity/quality of the ecosystem service affect the costs of production for the final good, the supply and demand for the final good, and the supply and demand for other factors of production. This information is used to link the effects of changes in the quantity/quality of the service to changes in consumer surplus and producer surplus, from which the economic benefits/costs can be derived.

The method is most easily applied in two specific cases. First, when the ecosystem service is a perfect substitute for other inputs, a change in quantity or quality of the service can be derived from changes in costs for the other inputs. For example, when land and water are prefect substitutes in the production of fruit, the value of a change in the ground water levels can be estimated by the changes in the costs of land required to keep production at the same level. Second, when the market price of the final product remains unchanged, consumer welfare is not affected, and only producers of the final good benefit from changes in quantity or quality of the resource. The value can then be estimated from changes in producer surplus only. This may occur in perfectly competitive and sufficiently large markets, in which changes in supply in one region do not affect the global market price.

An **advantage** of the production function method is that in theory it is well-suited to value ecosystem services since it is based on the notion that ecosystem services and economic benefits are strongly linked.

A **disadvantage** is that in practice the method is technically difficult to apply and has substantial data requirements. Care must furthermore be taken that the market value of other inputs in the production process are taken into account, so as not to overstate benefits from ecosystem inputs. Finally, the method is limited to valuing those ecosystem services that can be used as inputs in production of marketed goods. A possible exception to this is when dose-response functions are also considered to be production functions. For example, an increase in air pollution leads to increased morbidity and mortality. In this case a dose-response function is used to derive the effects of increased air pollution on health, and market prices and/or stated preference methods are used to estimate the costs of unit increases in morbidity and mortality.

Example Box 2: The value of mangroves for fisheries in Campeche, Mexico

Shrimp fisheries in Campeche, Mexico, account for approximately one-sixth of Mexico's total shrimp fisheries. The mangroves in the Laguna de Terminos are considered to be the main breeding ground and nursery habitat for these shrimps. The mangrove area was estimated at approximately 860 km^2 in 1980 and around 835 km^2 in 1990 – a loss of 2 km^2 per year. In order to analyse the contribution of mangroves to shrimp production, a shrimp production function is estimated with mangrove area and labour efforts as input factors, using production and input data from 1980 to 1990. The results show that a decline in mangrove area of 1 km2 causes a decline in shrimp harvest of about 14.4 metric tons and a loss in revenues of US$ 140,000 each year.

Source: Barbier and Strand (1998)

2.2.3 Cost-based methods

The damage cost avoided, replacement cost, and substitute cost methods are related methods that estimate values of ecosystem services based on either the costs of avoiding damages due to lost services, the cost of replacing environmental assets, or the cost of providing substitute services. The damage cost avoided method uses either the value of property protected as a measure of the benefits provided by an ecosystem. For example, if a wetland protects adjacent property from flooding, the flood protection benefits may be estimated by the damages should a flood occur multiplied by the probability of flooding without the wetland. The replacement cost method considers the cost of replacing an ecosystem as an estimate of its value (Freeman, 2003). In the substitute cost method the cost of ecosystem service substitutes are used as an estimate of the ecosystem service values. For example, the value of a wetland as a natural water reservoir may be approximated by using the costs of building and using an similar artificial reservoir. Basically, these methods assume that the amount of money society spends to replace an environmental service is roughly equivalent to the lost benefits that the service provides to society.

These methods all require the same first step, i.e., an assessment of ecosystem service(s) provided. This includes specifying the service(s), how they are provided, to whom they are provided, and the level(s) provided. The second step for the avoided damage cost method is to assess the potential physical property damages, and the final step is to derive the value of potential property damages. The second step for the replacement and substitute cost methods is to identify the least costly alternative means of providing the service(s), and the third step is to calculate the cost of the substitute or replacement service(s).

An **advantage** of cost-based methods is that they are relatively simple and inexpensive to apply. They do not require the use of detailed surveys or complex analysis. They provide surrogate measures of value that are as consistent as possible with the economic concept of use values, for services that may be difficult to value by other means. In addition these methods integrate well with the types of economic analysis that are often used in reality, such as cost-effectiveness analysis.

The methods also have rather severe **disadvantages**. First, they do not produce a strictly correct measure of economic value. The measures produced are not based on people's preferences for the ecosystem service, but on the assumption that the service's value is equal to the cost of replacement or damages avoided. In short, the supply side is used as an estimate for the demand side. The leads to a circularity in reasoning. For example, when a government decides not to replace a wetland that has disappeared, the conclusion from a cost-based point of view is that the estimated value of the service is low, while potential benefits from a demand point of view may be substantial. Also it leads to an inconsistent estimation of value. For example, the higher the initial level of ecosystem quantity/quality, the higher are the unit costs of replacing or avoiding a decrease in quantity/quality. From a cost-based point of view the value of such a decrease is high, while from a demand point of view the value of the decrease may be low, considering the initial levels were already high. Therefore, benefit measures that are based on costs will at some point incorrectly justify additional improvements in ecosystem quantity/quality. Also, reduction measures will in most cases reduce only part of the negative effects, implying that *ceteris paribus* the estimates produced by these methods likely understate the true values. Moreover, it is often difficult to find exact replacements for ecosystem services that provide an identical level of benefits. Similarly, substitute goods are unlikely to provide the same types of benefits as the original goods.

Given the severe limitations and inconsistencies of costs as an indicator ecosystem service values, valuation exercises should always aim to apply welfare based (benefit) valuation methods first, and use cost-based methods only in last resort.

Example Box 3: Value of mangroves for coastal protection

The coastal protection provided by mangroves in Southern Thailand has been valued using the substitute cost method. An important ecological function of mangroves is to serve as a windbreak and shoreline stabiliser. The value of this service has been estimated by calculating the cost of substituting this mangrove function by constructed breakwaters. The unit cost of constructing breakwaters to prevent coastal erosion is estimated to be around US$ 875 per metre of coastline. Based on ecological studies, it is considered necessary to preserve mangrove forests with a width of at least 75 meters along the coastline to stabilise the shore to the same degree as breakwaters. Given the above per-unit cost of breakwater construction, and assuming that a breakwater is 1 meter wide, the value stand of mangroves with a width of 75 meters is approximately US$ 116,667 per hectare.

Source: Sathirathai and Barbier (2001)

2.3 Revealed preference methods

Many ecosystem services, like aesthetic and recreation services, are not traded on markets and are not used explicitly as inputs in production processes with a final product that is traded on markets. However, the prices people are willing to pay in markets for related goods can be used to estimate their values. The two most important methods that make use of this notion are the hedonic pricing and the travel cost methods. We discuss these methods in detail below.

2.3.1 Hedonic pricing method

The hedonic pricing method measures the *implicit* price of an ecosystem service that is not traded on a market, as revealed through the *observed* price of a product that is traded on markets (Rosen, 1974). For this, many observations are needed on a product that is homogeneous in most respects, but different regarding a certain environmental characteristic, e.g., noise from traffic, vicinity of a park, etc. The difference in the sales prices of these two commodities can be used to derive the revealed willingness to pay for the ecosystem service. The hedonic pricing method may be used to estimate economic benefits or costs associated with environmental quality (e.g., air pollution, water pollution, noise), and environmental amenities (e.g., aesthetic views, proximity to recreational sites). Usually, house prices are used for valuing ecosystem services, although wages can be used as well, for example to value risks associated with working with environmentally hazardous goods.

In the case of house prices, these are related to the characteristics of the house and the property itself, the characteristics of the neighbourhood and community, and environmental characteristics. Thus, if non-environmental factors are controlled for, then any remaining differences in price can be attributed to differences in the ecosystem service under investigation. For example, if all characteristics of houses and neighbourhoods throughout an area were the same, except for the level of air pollution, then houses with better air quality would cost more, assuming people are aware of the difference. Higher sales prices thereby reflect the value of cleaner air to people who buy houses in the area.

To effectively apply the hedonic pricing method, information is required on the ecosystem service under investigation, along with data on property values and property and household characteristics for a well-defined market area. The sample should include houses with different levels of environmental quality, or different distances to an environmental amenity, such as open space or the coastline. The data are analysed using regression analysis, which relates property prices to property characteristics (also including neighbourhood, amenities, etc.) and the environmental characteristic(s) of interest. Thus, the effects of different characteristics on price can be estimated. The regression results reveal the change in property values due to a small change in the quantity or quality of an ecosystem service, *ceteris paribus*.

This method has the **advantage** that it makes use of revealed preferences; the estimation is based on data from an implicit but real market. If data are readily available, it can be relatively inexpensive to apply. If data must be gathered and compiled, the cost of an application can increase substantially.

There are several important **disadvantages** of the method (Hanley and Spash, 1993). One of the most important disadvantages is that hedonic price estimates measure a WTP for a non-marketed commodity at the location of investigation only, usually people's homes; the WTP for that particular service (e.g., noise or vicinity of a park) at other locations (work, leisure) is not measured. In contrast, the stated preference method does not have this particular problem. Second, the method assumes that the housing market is in equilibrium, but because of the large transaction costs involved in buying a house, equilibrium may be reached only in the medium to long term. Third, people may not be aware of the impacts of an environmental change. For these two reasons the method is more suited for estimating long-lasting environmental impacts. Other setbacks of the method are possible omitted variable bias (especially with hedonic house price functions it may be difficult to control for all factors influencing house prices), multi-collinearity (if serious, collinearity can cause unreliable and unstable parameter estimates) and market segmentation (different coefficients and thus different valuation of house characteristics by different groups of people). Also the method may be difficult to apply for valuing ecosystem services in poorly documented areas due to its large data requirements.

Example Box 4: Values of national wildlife refuges near urban areas on the US east coast

Wildlife refuges provide important cultural ecosystem services to especially urban populations. A hedonic pricing function was estimated using almost 88,000 property transactions in the year 2000 near 59 refuges along the US east coast. Results show that distance to a refuge has a negative effect on house prices and that this effect decreases when distance increases. More specifically, houses that are very close to a refuge (within half a mile) have substantially higher values than houses farther away; the mark-up is 3-10% depending on the scarcity of open space and the region. On average the net present cultural value (e.g., recreation, aesthetic) and possibly also health values (e.g., cleaner air) of refuges along the US east coast is around US$ 11 million per refuge.

Source: Yoo et al. (2014)

2.3.2 Travel cost method

A well-known method for valuation of recreation benefits is the travel cost method (TCM). This method is mostly used to estimate the economic benefits of a recreation site, based on an observed travel pattern of people who visit that site. It can also be used to derive recreation benefits that are related to changes in the quality of the environmental area, as long as these changes are captured by the number of visits to the environmental area. The principle of the TCM is inferring the value of recreation services by analysing revealed consumer behaviour on the transport market. The underlying premise is that the travel expenses that people incur to visit a recreation site represent the implicit price of access to the site. Travel expenses include the actual travel costs (e.g., the costs of using public transport, petrol and maintenance costs for travel by private car, cost of aeroplane tickets, etc.), time costs, and admittance fees. For example, to estimate the economic value of the recreation service provided by a forest, information is required on the number of people that visit the site, and on the time and costs they spend travelling to reach the forest. To successfully apply the travel cost method information is required on a large number of issues, i.e.:

- Number of visits;
- Demographic information of respondents;
- Substitute sites that the person might visit instead of this site;
- Travel distance to the site and to substitute sites;
- Perceptions of environmental quality of the site and of substitute sites;
- General characteristics of the site and of substitute sites;
- Value of travel time or the opportunity costs of travel time;
- Exact travel expenses;
- Amount of time spent at the site;
- Other locations visited during the same trip and amount of time spent at each.

An **advantage** of the travel cost method is that it applies well-accepted economic techniques for measuring value, and that it uses information on revealed rather than stated behaviour and preferences. It is based on the simple and well-founded assumption that total travel costs can be used to derive recreational value.

A **disadvantage** of the method is that in order to be effective and reliable it requires a relatively large data set and a large amount of information for each respondent. Data are usually collected through visitor interviews and questionnaires, which require sampling to cover different seasons or times of the year, and to ensure that various types of visitors from different locations are represented. Complex statistical analysis and modelling are required in order to derive the required recreation values. As a result, travel cost surveys are typically expensive and time consuming to carry out.

There are several additional sources of complication. First, defining and measuring the opportunity cost of time, or the value of time spent travelling and spent at the site, is generally problematic. Because the time spent travelling could have been used in other ways, it has a so-called opportunity cost, e.g., a person’s wage rate, or some fraction of the wage rate. These time costs should be added to the travel cost, or the value of the site will be underestimated. However, there is no strong consensus on the appropriate measure of opportunity costs, and the value chosen can have a large effect on benefit estimates. In addition, if people enjoy the travel itself, then travel time becomes a benefit, not a cost, and the value of the site will be overestimated. Second, the most simple models assume that individuals take a trip for a single purpose, i.e., solely to visit a specific recreational site. If a trip has more than one purpose in reality, the value of the site may be overestimated. It can be difficult to distribute the time and travel costs among the various trip purposes. Third, the availability of substitute sites will affect values. For example, if two people travel the same distance, they are assumed to have the same value. However, if one person has several substitutes available but travels to this site because it is preferred, this person’s value is actually higher. Some of the more complicated models account for the availability of substitutes. Fourth, people who highly value certain sites may choose to live nearby. If this is the case, they will have low travel costs, and therefore their high values for the site are not captured by the travel cost method.

Example Box 5: Recreation value of the Forest of Dean in the UK

The forest of Dean is around 90 km^2, located in a rural area with excellent facilities, and considered to be one the most attractive recreational forests in the region. In a travel cost valuation study the number of forest visits were estimated as a function of travel costs, visitors' trip characteristics, and visitors' individual characteristics. Data collection was done on-site and resulted in 199 useable observations. Model results show that travel costs have a negative and statistically significant effect on the number of visits. Consumer surplus was estimated at £5 per trip. Total consumer surplus for a larger area with a total of 40 million forest visits per year, was estimated at (40 million × £5 =) £200 million per year.

Source: King and Fraser (2013)

2.4 Stated preference methods

The methods discussed in the previous two sections have in common that they directly or indirectly rely on observed market behaviour. This has the clear advantage that the derived ecosystem service values are based on revealed preferences and actual behaviour. It has the inherent disadvantage that these methods do not capture actual and potential future changes in ecosystem services. Also the methods only capture values related to actual use of ecosystem services, and do not address potential non-use values (Hanley et al.., 1997). Methods that are specifically suited to deal with these two issues are called stated preference methods, i.e., methods in which people state their preferences or behaviour in hypothetical situations. The two most important stated preference methods are the contingent valuation method and the choice experiment method. We discuss these methods in detail below.

2.4.1 Contingent valuation method

The contingent valuation (CV) method can be used to estimate economic values for all types of ecosystem services. The term "contingent" denotes that valuation is based on a specific hypothetical scenario and description of the environmental service. The involves directly asking people for their maximum willingness to pay for a positive change in an ecosystem service, or for their minimum willingness to accept a negative change in an ecosystem service. For example, in the case that a wetland provides habitat for a popular species of animal, respondents to a survey might be asked to state how much additional tax they are willing to pay to preserve the wetland in order to avoid a decline in the population of that species.

The underlying premise of the method is that a hypothetical, yet realistic, market for buying or selling the use and/or preservation of an ecosystem service can be described in detail to an individual, who then participates in the hypothetical market by responding to a series of questions. These questions relate to a proposed change in the quality or provision of the good or service. The responses to these questions are then analysed to estimate the average value the respondents associate with the proposed change. This value can subsequently be aggregated over the affected population to derive a measure of total benefit (or cost). The basic steps in applying the contingent valuation method are:

- Define the valuation problem: This includes selecting the ecosystem services that are to be valued, and defining the relevant population is.
- Design the survey. This involves a number of steps including deciding what type of survey will be used (mail, telephone, face-to-face), the question format, payment vehicle, the WTP/WTA question, and pre-testing.
- Implement the survey: This includes selecting the survey sample, which in most cases should be a random sample from the relevant population.
- Analyse the results. This includes cleaning the data and dealing with non-responses to the survey and protest bids. Mean WTP per person should be calculated from the cleaned data – and extrapolated to the relevant population in order to derive a total value for the ecosystem service under investigation.

Most CVM studies are conducted via face-to-face interviews, web-based or postal surveys with individuals, but sometimes interviews are conducted with groups. A variety of question formats are used in order to elicit respondents' statement of their WTP/WTA for particular changes in the provision of ecosystem goods or services. The two main question formats used in CVM studies are:

- *Dichotomous choice*: respondents are presented with a bid amount and asked whether or not they are willing to pay/accept it. In the so-called 'double bounded' dichotomous choice format, respondents are presented with a second bid amount and again asked if they are willing to pay/accept, thereby establishing a range for the WTP/WTA (although the range is very broad when both bids are accepted);
- *Closed-ended choice card*: multiple bids are presented and respondents are asked to choose the bid that is closest to their maximum willingness to pay or their minimum willingness to accept;
- *Open-ended*: no bids are presented and respondents are asked to state how much they are willing to pay or accept.

An **advantage** of CVM is that it can be applied to estimate values for all types of environmental goods and services, including non-use values and also changes in ecosystem services that have not (yet) occurred. Because contingent valuation does not rely on actual markets or observed behaviour, it can in theory be applied to any situation, good or service.

A **disadvantage** of the method is that responses to willingness to pay questions are hypothetical and may not reflect true preferences. Hypothetical scenarios described in CV questionnaires might be misunderstood or found to be unconvincing to respondents, leading to biased responses. The most common forms of bias are related to strategic behaviour, survey design, payment instrument, anchoring to the bid amount starting point, and protest responses. It is important to carefully design and pre-test CV questionnaires in order to avoid or mitigate these biases. Another disadvantage of the contingent valuation method is that it requires complex data collection and sophisticated statistical analysis and modelling. The large-scale surveys that are necessary for contingent valuation can also be expensive and time consuming to conduct.

Example Box 6: Contingent valuation for protected coral reefs in the Philippines

This case study explores the demand by local and international divers for dive trips to protected coral reef areas in the Philippines. A small scale survey was carried out among diving tourists on and near Anilao, Mactan Island, and Alona Beach during the summer of 1997. The questionnaire used the following CV question: “How much would you be willing to pay as a daily, per person entrance fee to a marine sanctuary where fishing is prohibited, in addition to the other costs of the trip? US$ 0, US$ 1, US$ 3, US$ 5, US$ 10, other (please specify).” The results show a positive willingness to pay among divers to enter marine sanctuaries. Estimated annual potential revenues range from US$ 0.85-1 million on Mactan Island, from US$ 95-116 thousand in Anilao and from US$ 3.5-5.3 thousand on Alona Beach.

Source: Arin and Kramer (2002)

2.4.2 Choice experiment method

The choice experiment (CE) method is similar to the CV method in that it can be used to estimate economic values for virtually any ecosystem good or service. It is also a hypothetical method – it asks people to make choices based on a hypothetical scenario. The CE method is based on the idea that any good can be described in terms of its attributes or characteristics. Changes in attribute levels essentially result in a different good, and choice modelling focuses on the value of such changes in attributes. Values are inferred from the hypothetical choices or trade-offs that people make between different combinations of attributes. The CE method is different from the CV method in that it asks respondents to select between a set of alternatives, rather than asking directly for values. Values are derived from the responses by including a so-called payment vehicle (e.g., price of the good) as one of the characteristics.

The CE method addresses a number of the difficulties associated with the CV method. For example, rather than simply asking respondents how much they are willing to pay for a single improvement in a given non-market good, a choice model forces respondents to repeatedly choose between complex, multi-attribute options. In a typical CE study respondents are presented with a series of choices that each consist of two or more choice options. The choice options are described using a common set of attributes, which summarise the important aspects of the options. Often the status quo is included as a choice option. For each choice a respondent evaluates the different choice options and chooses his or her preferred option.

Because the CE method focuses on trade-offs among alternatives with different characteristics, it is especially suited to policy decisions where a set of possible actions might result in different impacts on ecosystem services. For example, a restored wetland will improve the quality of several services, such as floodwater storage, drinking water supply, on-site recreation, and biodiversity. Similar to the CV method, the basic steps in applying the CE method are:

- Define the valuation problem: This includes defining which ecosystem services are to be valued and what the relevant population is.
- Design the survey. This involves a number of steps including deciding what type of survey will be used (mail, telephone, face to face), determining the choice set (i.e., what characteristics will respondents be required to choose between), choosing the payment vehicle (the monetary attribute), and pre-testing. Ideally, focus groups followed by pre-testing should be used to set and test the relevant levels of the choice attributes used.
- Implement the survey: This includes selecting the survey sample, which in most cases should be a random sample from the relevant population.
- Analyse the results: The statistical analysis for contingent choice is generally more complicated than that for contingent valuation and requires the use of statistical analysis and specifically choice models to infer willingness to pay from the trade-offs made by respondents. The average WTP for each of the attributes is estimated is extrapolated to the relevant population in order to calculate a total value for the ecosystem service under investigation.

An **advantage** of the CE method is that it is an efficient means of collecting information, since choice tasks require respondents to evaluate multi-attribute profiles simultaneously. In addition, economic values are not elicited directly but are inferred by the trade-offs respondents make between monetary and non-monetary attributes. As a result, it is less likely that Willingness to Pay (WTP) information gathered using this method will be biased by strategic response behaviour. A further advantage of the choice model approach is that research is not limited by pre-existing market conditions, since the levels used in a choice experiment can be set to any reasonable range of values. As such, the choice modelling is useful to use as a policy tool for exploring proposed or hypothetical futures or options (for example, in a decision support tool based on the results). Finally, and perhaps most importantly in the context of non-market valuation, choice experiments allow individuals to evaluate non-market benefits described in an intuitive and meaningful way, without being asked to complete the potentially objectionable task of directly assigning dollar figures to important values such as culture.

A **disadvantage** of the method is that, similar to the CV method, the choices made are hypothetical and may not reflect true preferences or behaviour. It is therefore important to carefully design and pre-test CE questionnaires in order to avoid or mitigate hypothetical bias. Another disadvantage is that the method requires complex data collection and sophisticated statistical analysis and choice modelling. The large-scale surveys that are necessary can also be time-consuming and expensive. Moreover, the time needed to explain the attributes and attribute levels and what is required from a respondent in a CE, along with the generally large number of choices with which a respondent is confronted, make that there is a risk of respondent fatigue, potentially leading to a reduction in accuracy of the results.

Example Box 7: Local willingness to pay for coral reef conservation in Guam

Guam's coral reefs provide important cultural, recreational, and non-commercial fishery values that are not easy to measure using direct market methods. However, it is extremely important to include non-market values in economic assessments to ensure that governments and policy makers are aware of the full value associated with natural assets such as coral reefs.

The choice experiment investigated three important non-market benefits associated with Guam's coral reefs: local recreational use, abundance of culturally significant fish species, and non-commercial fishery values. In addition, pollution and reef fishery management were also included as attributes in the choice experiment, because they affect the health of reefs. The pollution attribute measured preferences for controlling land-based sources of pollution (including sedimentation, run-off, and sewage outflow), while the reef management attribute measured preferences for eliminating destructive fishing practices. Income tax was included as the payment vehicle in the choice experiment.

The results of the choice model indicate that significant economic values are associated with the three non-market benefits included in the survey. Guam's residents appear to place a similar value on the reefs' ability to provide local recreational benefits and supply culturally significant fish species. In addition, the results indicate that maintaining reef fish and seafood stocks at a level that can support the culture of food sharing is very important. The WTP for sufficient fish catches to share with family and friends was valued at US$ 92 per fisherman per year.

Source: Van Beukering et al. (2007)

2.5 Meta-analysis and value transfer

Often value estimates for ecosystem services are obtained by applying existing value estimates, rather than through applying a valuation method in a specific situation. Methods that are closely interlinked are meta-analysis and value transfer. We discuss these methods below.

2.5.1 Meta-analysis

Meta-analysis is a method with which a researcher can summarise, synthesise and analyse the available empirical evidence on a certain topic, e.g., ecosystem service value estimates from studies that employ one or more of the previously discussed valuation methods. The procedure of doing a meta-analysis is relatively straightforward. First, the available empirical evidence on the topic of interest is gathered by using relevant predefined keywords in standard available search engines (Google Scholar, Scopus, Web of Science, etc.). Second, the outcomes of studies and their the characteristics are coded and put into a database. Third, additional characteristics of studies and the areas in which the values are obtained from external resources. For meta-analyses on non-market valuation studies these generally include income of the relevant population, population density of the relevant area, land use in the relevant area, and data on the supply and quality of the ecosystem service that is being studied. Finally, the data are analysed using more or less advanced statistical techniques, such as regression analysis, and insights into the relevant sources of variation in ecosystem service values are obtained. Interesting empirical applications of meta-analysis in the field of ecosystem service valuation are, among others, Bateman and Jones (2003), Brander and Koetse (2011), Brander et al. (2006), Johnston et al. (2006), Van Houtven et al. (2007) and Woodward and Wui (2001).

A clear **advantage** of meta-analysis is that pooling the estimates from various studies may provide a preferable estimate of value, i.e., an estimate with a smaller confidence interval, and provides quantitative insight into which factors are relevant in explaining the variation in the available empirical evidence. Since the studies used in the meta-analysis are as a rule based on different data-sets from multiple countries and different time frames, meta-analysis generally provides greater possibilities for generalisation than a single case study does. Also the development of Geographical Information Systems (GIS) allows for gathering spatially specific case-study data. By including these data in the model specification more spatially explicit predictions and generalisations can be made.

A **disadvantage** of meta-analysis is that is generally time consuming and that the researcher is dependent on the availability of empirical evidence. Looking for existing meta-analyses on a specific topic is therefore advisory. Another disadvantage of meta-analysis compared to individual case-study analysis is that it is generally excludes case-study specific features. The reason is that in a meta-analysis the outcomes and characteristics of studies have to be made comparable, and specific features of individual studies can often not be included in the final model specification. When using value transfer for obtaining an estimate of an unstudied ecosystem service in a specific area, this less of a problem. However, when the ecosystem service under investigation has (very) specific features that are deemed to be relevant for its value, using a meta-analysis value function for obtaining a value estimate may severely over- or underestimate the true value. In this case it may be advisable in a value transfer to use a case study that resembles the specific situation more closely, rather than using the outcomes of a meta-analysis.

2.5.2 Value transfer

The value of ecosystem services at one location can be estimated based on the results of valuation studies of environmental services at other locations, thereby transferring values from one site to another. This technique is called 'value transfer', and can be used for both benefits and costs. In the literature, value transfer is commonly defined as the transfer of monetary environmental values that are estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the changes in physical characteristics and their monetary values. The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making. The decision of whether to undertake an original study or to use existing value estimates can be considered in terms of the acceptability of errors produced by value transfer and the level of precision sought.

Although value transfer is used extensively in practice, relatively little published evidence exists about its validity and reliability. The range of transfer errors found in the empirical literature is large (see, e.g., Brouwer and Bateman, 2005; Rosenberger and Loomis, 2003; Van den Berg, Poe and Powell, 2001). Stated differently, the value predicted by a value transfer exercise can largely over- or understate the true value. Three important sources of error can be distinguished: (1) the error incurred when estimating the original unit value, (2) the error incurred when transferring the original unit value to the new policy context, (3) the error incurred when aggregating the transfer unit value to the whole population of beneficiaries and calculation of the total economic value (see also Brouwer, 2000; Rosenberger and Loomis, 2003). The potential magnitude of transfer errors have to be considered in the light of the purpose for which the user (policy or decision maker) wishes to use previous valuation results. In some cases a transfer error of 50 percent may be considered too high, in other cases such an error may be acceptable.[5] The acceptability of the error will depend on subjective judgement by the user, the purpose and nature of the evaluation (e.g., costs-benefit analysis, pricing/cost recovery, environmental liability) and the phase of the policy cycle in which the evaluation is carried out.

[5] Note that using market analysis and market prices for value transfer may also lead to large transfer errors. Although prices are observed, value transfer makes use of demand curves which have to be estimated and may contain substantial estimation error. Also, it's not a given that primary studies give better estimate – a value transfer based on a good meta-analysis might be statistically better than a primary study with a smallish sample, or bias/rejection problems.

Several necessary conditions should be met to perform effective and efficient value transfers (Desvousges et al., 1992). First, the policy context should be thoroughly defined, including:

- Identifying the extent, magnitude, and quantification of expected site or resource impacts from the proposed policy action in both quantity and quality terms.
- Identifying the extent, spatial distribution and characteristics of the population that will be affected by the expected site or resource impacts.
- Identifying the data needs of an assessment or analysis, including the type of measure (unit, average, marginal value), the kind of value (use, non-use, or total value), and the degree of certainty surrounding the transferred data (i.e., the accuracy and precision of the transferred data).

Second, the study site data should meet certain conditions for critical value transfers:

- Studies transferred must be based on adequate data, sound economic method, and correct empirical technique.
- The study contains information on the statistical relationship between benefits (costs) and socioeconomic characteristics of the affected population.
- The study contains information on the statistical relationship between the benefits (costs) and physical/environmental characteristics of the study site.
- An adequate number of individual studies on a recreation activity for similar sites have been conducted in order to enable credible statistical inferences concerning the applicability of the transferred value(s) to the policy site.

Third, the correspondence between the study site and the policy site should exhibit the following characteristics:

- The environmental resource and the change in the quality and/or quantity of the resource at the study site and the resource and expected change in the resource at the policy site should be similar. This similarity includes the quantifiability of the change and possibly the source of that change.
- The markets for the study site and the policy site are similar, unless there is enough usable information provided by the study on own and substitute prices. Other characteristics should be considered, including similarity of demographic profiles between the two populations and their cultural aspects.
- The conditions and quality of the recreation activity experiences (e.g., intensity, duration, and skill requirements) are similar between the study site and the policy site.

Thus, while value transfer may provide a quick and cheap alternative to original valuation research, some conditions must be met if it is to provide reliable results (for best practices and guidelines, see, e.g., Bateman et al., 2009, 2011). Above all, the local circumstances and conditions in the new decision-making context need to be closely related to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explanatory variables into the transfer, as is usually the case when meta-analysis results are used for value transfer. However, this also increases the data requirements and the complexity of the analysis. Also, the potential for conducting a sound and reliable value transfer hinges on the number, quality and diversity of valuation studies available. The larger, the better and the more diverse the existing set of studies is, the more reliable results of a meta-analysis will be, or the higher the probability that there is a study that closely resembles the policy site.

The methods used for value transfer can be broadly categorised into two types (Navrud and Ready, 2007). The simplest approach is to attempt to find study sites which appear similar to the policy site and transfer mean values from the former to the latter (e.g., Muthke and Holm-Mueller, 2004). Such 'univariate' transfers are frequently used in practical decision making, but are crucially dependent upon differences between transfer sites. Clearly all sites are to some degree dissimilar (e.g., unique ecosystem habitats or the spatial pattern of substitutes around a site is unique), and it is the degree to which this dissimilarity affects values that will determine the appropriateness of univariate or mean value transfers. It is because of such concerns that value function or 'multivariate' transfer approaches have been developed. Here statistical techniques are used to estimate value functions from study site data. These are then used to predict new values for policy sites. This is achieved by assuming that the underlying utility relationship embodied in the parameters of the estimated model applies not only to individuals at the study sites but also to those at policy sites. Usually, these parameters are kept constant, while the values of the explanatory variables to which they apply are allowed to vary in line with the conditions at the policy site. In those cases where results are used from studies carried out years ago, an important question obviously is to what extent preferences and parameters have changed (Brouwer, 2006).

Pearce et al. (1994) argue that because value function transfers allow the analyst greater control over differences across sites, they should in principle yield lower transfer errors than simple mean value transfers. However, empirical evidence regarding this assertion is mixed with sometimes the opposite result being observed (e.g., Bergland et al., 1995; Barton, 2002; Ready et al., 2004). This is partly due to the lack of a systematic assessment of a set of (theoretically driven) baseline conditions needed to be in place for valid and robust value transfer. In a cross-country comparison, Brouwer and Bateman (2005) show that a simple unadjusted unit value transfer works best for similar case study sites, while errors generated by simple mean value transfer are considerably larger than those arising from function transfer across dissimilar case study sites. As expected, if conditions are not the same across study and policy sites, some degree of adjustment helps reducing the error.

Example Box 8: The economic value of the World's wetlands

Value transfer has been used to estimate the economic value of the World's wetlands. Using 246 separate observations of wetland value from 89 studies, a value transfer function was estimated. Wetland values have been reported in the literature in many different metrics, currencies and refer to different years (e.g., WTP per household per year, capitalized values, marginal value per acre, etc). In order to enable comparison, these values have been standardized to US$ 2000 per hectare per year. This standardization included a *purchasing power parity (PPP)* conversion in order to account for different price levels in different countries. The average annual wetland value in this data set is just over US$ 3,000 per hectare. The median value, however, is US$ 170 per hectare per year showing that the distribution of estimated values is skewed with a long tail of high values.

The value transfer function was estimated by computing a functional relationship between the standardized wetland values and a number of important explanatory variables, including wetland type, income per capita, population density, wetland size and continent. Given information on the same characteristics of other wetland sites that are of policy interest, this estimated value function could then be used to predict the value of those wetlands. Values were transferred to around 3,800 wetland sites around the world to estimate the global economic value of wetlands. The total economic value of 63 million hectares of wetland around the world is estimated at US$3.4 billion per year.

Source: Schuyt and Brander (2004)

2.6 Choosing a valuation method

In Table 1 we give an overview of the ecosystem services that are distinguished in the Millennium Ecosystem Assessment (2005), and the methods that are commonly used or can be used to value these services. In the table the value transfer method is not included, but note that it has been applied often for various types of ecosystem services. *Provisioning services* can generally be valued using direct market methods. Of course, when potential future changes in quantity and/or quality of these services are to be valued, non-market methods are possibly more suitable. The same holds for most *regulating services*. These can generally be valued through direct or indirect market methods, but hypothetical and potential future changes are also valued through non-market methods. Cultural services are usually not traded on markets, so direct market methods are not available. Hedonic pricing can and fairly often is used for valuing recreation and aesthetic services, while travel costs are generally used for recreation services only. Stated preference methods are most often used for valuing cultural services, however. Reasons are that these methods allow for valuation of hypothetical change and thereby for *ex ante* estimation of welfare changes due to physical changes that have not occurred before (or at least, not in that situation). Finally, *supporting services* may be valued through production functions. For example, values of biodiversity, land quality and water may be derived through food production functions.

Table 1. Ecosystem services and possible valuation methods (source: based on Table 6.1 in Koetse et al., 2015)

	Direct market methods [a]			Revealed preference methods [a]		Stated preference methods [a]	
	MP	PF	CB	HP	TC	CV	CE
Provisioning services							
Food	X	X				X [b]	X [b]
Fresh water	X	X				X [b]	X [b]
Wood and fibre	X	X				X [b]	X [b]
Fuel	X	X				X [b]	X [b]
Regulating services							
Climate regulation			X			X [b]	X [b]
Flood regulation			X	X		X [b]	X [b]
Disease regulation	X		X			X [b]	X [b]
Water purification	X	X	X			X [b]	X [b]
Cultural services							
Aesthetics				X		X [c]	X [c]
Recreation				X	X	X [c]	X [c]
Education						X [c]	X [c]
Spiritual						X [c]	X [c]
Supporting services							
Nutrient cycling		X	X				
Soil formation		X	X				
Primary production		X	X				

[a] MP=market price method; PF=production function method; CB=cost-based methods; HP=hedonic pricing method; TC=travel cost method; CV=contingent valuation method; CE=choice experiment method.
[b] Although markets exist for most provisioning and regulating services, valuation of hypothetical changes and changes that have not yet taken place may also and perhaps are preferably be done though stated preference valuation methods.
[c] For most cultural services markets do not exist, implying valuation of both current and hypothetical situations and changes can often only be done though stated preference valuation methods.

Some of the general points to consider when choosing a method for the valuation of ecosystem goods and services are related to the following issues (leaving aside the fact that some methods are more expensive and time-consuming than others):

- *Type of ecosystem service to be valued* – some methods are more appropriate than others for specific goods or services.
- *Type of economic value to be estimated* – whilst use values are estimated by all of the various techniques, non-use values can only be estimated by stated preference methods.
- *The purpose of the valuation* – some studies require valuation methods based on the estimation of marginal values, whilst other studies require the estimation of total economic value, given by consumer (or producer) surplus.
- *Data and information availability* – existence and availability of data.

- *Accuracy of results required* – the degree of uncertainty surrounding the outcome of different methods varies widely, and may be decisive when choosing a specific method for a specific purpose. For example, the acceptable level of uncertainty is much higher in a pre-feasibility cost-benefit analysis than when wanting to establish the 'correct' user fee for a certain ecosystem service based on current levels of cost recovery.

As described in the previous sections, all valuation methods have strengths and weaknesses, and a decision on which method to use in a particular situation requires experience and judgment on the part of the analyst. It is often possible to use more than one valuation technique and compare the results. All methods involve some uncertainty; if the analyst has multiple estimates, he or she will have greater confidence in the value of the proposed change. Several of the valuation methods typically use data from a household survey (for example contingent valuation, choice experiment and travel cost studies). When a method requires that primary data be collected with a household survey, it is often possible to design the survey to obtain the data necessary to undertake more than one valuation method (e.g., Alberini et al., 2007; Mogas et al., 2006, 2009; Stevens et al., 2000).

2.7 Discussion

Expressing the importance of ecosystem services into monetary values is increasingly popular among researchers and practitioners. The conversion of quantities into values has both advantages and disadvantages. An important advantage of estimating monetary values for ecosystem services is that this allows for a sensible comparison of costs and benefits of public and private decisions that affect the value of ecosystems and their services. Not having a value for ecosystem services makes that these comparisons are not possible, and may give the erroneous impression that there is no value at all. This leads to ongoing overexploitation of ecosystems, and to further loss of biodiversity and environmental degradation.

At the same time, a number of disadvantages of economic valuation of ecosystem services are evident, in particular with some of the non-market valuation methods. For example, stated preference approaches typically rely on hypothetical representations of ecosystem service change and hence elicit hypothetical WTP or WTA, not what people actually pay, which is considered a more reliable indicator of economic value. Also, stakeholders and especially the public at large, may not be aware or familiar with the range of ecosystem services provided at local and global scale, let alone that they have experience paying for these often non-priced public goods and services. This may result in valuation bias that has to be accounted for and tested in the design of the valuation study if possible. Moreover, the results of non-market valuation methods are often site- and context-specific, thereby reducing their general applicability. It is furthermore difficult to project values and preferences into the future, although this problem is not specific for economic value but holds for any assessment of value (also implicit value assessments).

Finally, economic valuation of public environmental goods is based on an individual approach. Alternative approaches have been developed over the past decades focusing on and interpreting environmental valuation more as a process of social construction, in which public preferences are constructed through social interaction and engagement. Here, in addition to economic values, socio-cultural values and perceptions play a crucial role in determining the importance of natural ecosystems to society and their preservation. Socio-cultural values are based on the notion that healthy ecosystems are a crucial source of cultural well-being and essential for a sustainable society (Norton, 1987). Ecosystem-related socio-cultural values are defined more broadly and include equity, physical and mental health, education, cultural diversity and identity (heritage value), freedom and spiritual values, which are more difficult to capture through the concept of WTP or WTA. Moral and ethical considerations often play a role here too, including the idea that not all public environmental goods and services are amenable to privatization and commercialization. People may behave differently as consumers when buying market goods and services than as citizens when addressing public environmental goods and services. These different approaches are not necessarily mutually exclusive, and may complement each other and as such enrich the underlying information base for policy and decision-making.

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3 Innovative methods to account for socio-cultural values

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3.1 Introduction

In neo-classical theory of consumer behaviour, decisions are based on individual rationality. Choice depends on preferences and preferences are assumed to be fixed, informed and to measure the utility that the individual attaches to a good's capacity to satisfy his or her self-interest.[6] Choices are articulated through a market place and follow a process whereby the individual trades off alternatives, the values of which are taken to be commensurable with one another.

This assumption of fixed and informed preferences is central to many of the key axioms behind consumer choice theory. It also provides the foundation for the economic valuation of environmental goods. However, while there are merits in having tractable explanations of behaviour, many economists would agree that actual decision-making is a more complex affair.

In the first instance, individuals may be directed by a perception of well-being that is informed by objectives other than the maximisation of their personal utility or income. Their decisions might be guided by various social norms, rules and considerations for others, or by personal ethical or spiritual beliefs. These considerations are not always commensurable with expressions of utility maximisation based on willingness-to-pay approaches, e.g., contingent valuation or choice experiments.

Both the nature of the good and the decision-making context are important. Environmental goods are typically not divisible to the interests of the individual. Many are public goods are non-excludable and so must be shared. As such, it is not possible to divorce one's own interests from others' needs and behaviour. Being non-rivalrouse, many public environmental goods, especially in modern times, are also vulnerable to excessive use or changes to the wider ecosystem. The

[6] See for example Bernheim, B.D., Rangel, A., 2007. Behavioral Public Economcis: Welfare and Policy Analysis and Nonstandard Decision-makers, in: Diamond, P., Vartiainen, H. (Eds.), Behavioral Economics and its Applications. Princeton University Press, Princeton, pp. 7-84.

implications of their use (or over-use) are uncertain and so an individual may not be fully informed of the consequences of his or her behaviour.

3.2 Disciplinary perspectives on values

3.2.1 Endowments and values

The observation that behaviour and choice frequently fail to conform to principles prescribed by economic theory has given birth to the discipline of behavioural economics. Psychology has been a key ingredient, most notably the highly citied paper by Kahneman and Tversky (1979) that described how individuals were averse to a loss compared with a comparative gain. A substantial literature has demonstrated this phenomenon and shows that the starting point is key to an understanding of what might be perceived as a loss or a gain (Kahneman, 2011). Existing endowments are one evident influence on the starting point. Individuals are frequently observed to be unwilling to trade away from their current allocation of goods or may wish to receive greater compensation in return. A related observation is that of a psychological attachment to their existing endowment or the status-quo (Masatlioglu and Ele, 2005; Samuelson and Zeckhauser, 1988). Indeed, an attachment to the status-quo could be a rational response where an alternative allocation could imply uncertainty, particularly where the individual is reasonably content with his or her present circumstances.

Endowments do not have to consist of physical possessions. They can also be a result of social and cultural norms. People hold a variety of social and cultural values (SCV) informed by their upbringing and the society in which they live. These values have a great influence on behaviour. Environmental philosophers have reduced these to three dimensions of human-environment relations, namely utility, ethics and aesthetics (O'Neill, 1993; Sagoff, 1994).

a) utility

Utility is the construct promulgated by economics to represent well-being, People are assumed to be informed and can choose between goods on the basis of the relative marginal utility that they provide. Expected utility is the focus of the choice as people are presumed to be consumers whose aim is to maximise their expected personal welfare based on the best outcome. This quest is most typically illustrated by the maximisation of personal income, but can extend to other motives that advance personal welfare including a consideration for others or for future generations.

b) ethics

Ethics are informed by culture and frequently relate to shared (rather than individual) social values. Behaviour may be guided by consequentialist (goals or end based) motivations or by deontological motivations relating to the expectations of rights and duties that form part of one's culture (Pearson et al., 2012). This sense of what is right and wrong - or what behaviour is acceptable - is often internalised within the individual's own value system, becoming inseparable from what that person might regard as their well-being. However, this does not inevitably imply ethical behaviour. One's perspective on equality cannot easily be divorced from the prevailing distribution of property rights, including those pertaining to the individual (Kumar and Kumar, 2008) .

c) aesthetics

Aesthetic values may coincide with ethics. In philosophy, aesthetic values are not confined to superficial art or beauty, but to the senses and emotions. These are influenced by spiritual or other beliefs that provide the individual with principles to aspire to. More precisely, Jax (2013) refers to eudaimonistic values that relate to what is understood to be the "good life". Quality of life is the goal to which most people aspire and, in its purest form, should equate to happiness as an aesthetic value. While this should be people's ultimate aspiration, the route markers are too commonly confounded by other more immediate motivations ranging from simple survival through satisficing objectives to income maximisation.

At a policy level, much of the justification for spending on the environment revolves around such familiar economic criteria as the contribution to local development, tourism, employment, rather than considerations of economic welfare. In fact, there is a narrower gap between utility based economic valuation and other methods that seek to incorporate a wider range of socio-cultural values. It is worth reminding ourselves of humankind's fundamental needs and values. Maslow's Hierarchy of Needs (1954) arranges these needs in a pyramid with basic physiological needs at the base followed by the three other basics needs of safety, love and belonging and esteem, succeeded by the higher need of self-actualisation (to realise one's potential) and, in subsequent writings, self-transcendance, i.e., a higher spiritual or altruistic goal. Ascending the pyramid requires the individual to satisfy each level of needs in sequence.

In the UK National Ecosystem Assessment, Church et al (2011) favour instead the related Human Scale Development Matrix of Max-Neef (1989, 1992). This consists of a matrix of four existential needs that have a long history in psychology[7], namely having, doing, being and interacting. The first of these includes material well-being, but also family, religion, work, education and health. Socio-cultural values attach to each need, including protection, affection, understanding, participation, leisure, creativity, identity and freedom.

[7] See, for example, Sartre (1943), Marx (1848) and Fromm (1956).

3.2.2 Economic and alternative values

Perceptions of well-being are informed by these socio-cultural values of utility, ethics and aesthetics. Indeed, utility, as an economic construct, does not have to be thought of as being distinct from other motivations. Neo-classical economics assumes that any element that the consumer finds important will enter the utility function (Arrow, 1963; Becker, 1993). If ethical, aesthetic or spiritual values are internalised within the decision-making process, then the individual will choose a path to his or her personal utility that satisfies these values. He or she will rationally trade-off the various motivations to arrive at the best decision. Cost-benefit analysis is compensatory in that in relies on an argument of additive utility by which beneficiaries can compensate the losers (Munda, 1996).

The contrary argument is that these alternative values are incommensurable, i.e., that they cannot be reduced to a single metric or traded against one another. Individuals can behave according to individual rationality or social rationality, but the ability to act consistently according to both is constrained. Economics tends to a logical or positivist approach, but social rationality acknowledges the influence of social or cultural institutions that help define individuals' constructs of the world and provide normative guidance on what should be. Thus classical institutionalism regards values and preferences as being grouped into classes between which there can be restrictions on trade-offs (Vatn, 2009).

Social rationality has a varying input on decision-making. Routine decisions may be based on simple rules, including utility maximisation exercised in the market (Hill, 2008). Sagoff (1988) accepts that consumer preferences are indeed relevant when individuals are behaving as consumers. Where uncertainty arises, decisions are made on the basis of expected utility and decisions are guided by either substantive or procedural rationality (Simon, 1979). The former implies conformance with an internal preference ordering irrespective of the manner in which the decision has to be made. However, rather than being fixed and informed, other choices might be procedural, i.e., the result of a process. This could involve the collection of relevant information, but it could also involve a role for the context in which the decision is made (Martinez-Alier et al., 1998; Muthoo, 1998). These competing sources of rationality mean that decision making can be very dynamic, decided by time, place and immediate past experience or invoked "on the spot" (Gowdy and Mayumi, 2001). As such, there may be a varying role for situational learning, habit, heuristics, inertia and endowments.

The wider external social and cultural context includes influences such as education and up-bringing, culture, property rights and social norms. Complex choices, for example decisions that have implications for others, may be more influenced by socio-cultural values. One characteristic is that individuals may behave as citizens rather than consumers (Sagoff, 1988). As a member of a community, they may take a moral or ethical position that coincides with their perception of the greater good. Of course, this is not always the case. There is no shortage of examples where people have behaved in ways that take no account of their neighbours. However, morals or ethics are more likely to play a part where there are inter-dependences with others and where communal or public goods are concerned.

Culture also attaches certain shared meanings to nature. These are translated by individuals based on everyday practice and the role of social institutions (Church et al., 2011). Where there is a dependence on common pool resources, social norms or more formal institutional arrangements have often evolved to govern their use. These arrangements, together with their de-facto rights and expectations of reciprocal transactions, are often the bedrock of shared cultural values and local social capital. They provide the context in which many decisions relevant to the environment are made.

Alternatively, rather than the instrumental values associated with use of the environment, the individual might adopt an ethical position that is ecocentric or biocentric, namely a position where nature is assumed to have rights too (Jax, 2013; van Steenbergen, 1994). A related position, relevant to environmental sustainability, is one in which rights are allocated to future generations (Dobson, 2003; Shelton, 1991).

Lexicographic preferences can be a consequence of all these phenomena. They occur when people decline to make trade-offs even when others do or where an external objective assessment suggests they should. Although utility theory acknowledges that people can often have a strong attachment to particular outcome, these types of responses regularly upset valuation studies and the expectation of a smooth utility function. They emerge when respondents appear to be unwilling to trade-off one good or utility state for another. Various factors could provoke people into refusing to trade, including an attachment to the status-quo, issues around property rights or the nature of the valuation method, However, one reason why a strong, non-negotiable position may be taken could be that an individual attaches incommensurable values to the goods being traded. These values should not be written off or ignored in any analysis of decision-making.

3.3 Values and ecosystem services

The economic perspective on values is represented by the typology of Total Economic Value (TEV). For ecosystem services, individuals most typically experience direct use value from provisioning services and indirect use value from regulating services. Many cultural ecosystem services are associated with use values, although most ecosystem services have nonmaterial or intangible dimensions (Chan et al., 2012). Socio-cultural values can be attached to any of these categories of TEV, but have a particular association with non-use values, including the utility one associates with bequest, vicarious and existence values that do not directly benefit the individual.

Economic values are often presumed to be associated with provisioning services because many (but not all) of these services are captured by markets. This is the context for trade-offs and monetary exchange. Even where markets do not exist, a case can be made to creating them or hypothesising them into existence. The indirect values associated with regulating ecosystem services often involve a shared common reliance on the resource. They can also involve a lack of awareness or uncertainty over the nature of the ecosystem service or of the underlying ecological processes. Non-use values are clearly influenced by socio-cultural context, but can be challenging to value. What may at first appear to be a non-use value, could in fact involve indirect use or "passive use" , in contrast to pure non-use. Indeed, some economists, e.g., Freeman (1993) express scepticism over the very existence of non-use values. If intrinsic values are defined as ‚the value of environmental elements in their own right, then these values do not belong in a typology which is only concerned with values of anthropogenic origin. However, intrinsic values do often enter into debates over the rights of nature.

What is evident from the TEV approach is that conventional valuation methods that measure utility-based motivations are most easily applied to provisioning services (Martin-Lopez et al., 2013), particularly where these are exchanged in the market. Other values, including those influenced by ethical and aesthetic positions, are relevant to all the TEV categories, but are especially influential in the valuation of non-use.

Contrary to the von Neumann and Morgenstern definition of expected value, uncertainty can also be a characteristic of context. Choices are subject to bounded rationality making utility maximisation a more challenging to achieve than a satisficing objective (Simon, 1979). Nature is subject to such uncertainty. It has evolved a high degree of resilience, but contemporary pressures mean that its status is often subject to a lot of uncertainty These pressures have often resulted from a limited understanding of the ecosystem (Winkler, 2006) which may respond non-linearly to exogenous change (Daily et al., 2000) and is subject to discontinuities and local equilibria. The response may also occur only after a time-lag that obscures marginal change and precludes the opportunity to attach marginal values (Sagoff, 1994). Alternatively, changes can occur suddenly and unexpectedly after a threshold has been reached. Regulating services, such as those that contribute to good water quality, are often poorly understood. They might provide indirect use values and, as such, attract modest attention compared with services that meet other anthropocentric values. In principle, a significant option value may, or may not be recognised, as being attached to the sustainability of these services.

3.4 Economic valuation

Economics assumes that ultimately all values depend on people's willingness to pay for a good is presumed to be guided by underlying preferences. It is these preferences - and therefore utility - that count in the efficient allocation of resources. In perfect markets, values are reflected in prices that are typically established by market exchange.

In practice, willingness-to-pay also depends on income or wealth. In addition, where conventional markets are not available, as in the case of many environmental and public goods, market places may either have to be created or hypothesised to inform policy and decision making through the elicitation of information on the values associated with alternative choices (Daily et al., 2000; USEPA, 2009).

Stated preference methods, such as the contingent valuation method (CVM) (introduced in Chapter x), aim to create a hypothetical market consisting of a choice between an environmental good and money (representing a loss or gain of income depending on the welfare measure used). Eliciting preferences in this way is vulnerable to many kinds of application failures, hypothetical biases and other biases. Various standards of best practice set out how surveys should be implemented to minimise the risk of error. In principle, people's stated preferences are assumed to include an introspective account of ethical and other values. Sometimes, the respondent's wish to overtly account for these values may be reflected in the very biases that the method is trying to avoid, such as strategic bias or the warm glow effect of allocating a high willingness-to-pay for a worthy cause, especially in what is inevitably a hypothetical scenario. On other occasions, respondents refuse to express a willingness-to-pay on the basis that the good (often an environmental good) should not be "priced" or "is without price". A well implemented applied survey will analyse respondents' reasons for giving such a response, but typically such responses are omitted from the calculation of total willingness-to-pay.

In the process of normal market exchange there is no guarantee that people take sufficient account of the range of values that they hold. The same is true, and perhaps more so, for respondents to stated preference methods. The nature of the exercise, by being focused on a willingness-to-pay, means that they might not have been motivated to do so. Economic valuation methods are - like any other method - value articulating institutions (Martin-Lopez et al., 2013; Vatn, 2009). In contingent valuation, the types of values sought are signalled to the respondent by willingness-to-pay question. For choice experiments, price is one or several attributes, but the question still assumes that the respondent is able to make a trade-off between the environmental good and money. If the respondent judges that it makes the question easier, the transaction cost of stating a willingness-to-pay or making this trade-off is reduced by omitting consideration of alternative values or uncertainty. However, social institutions are an essential foundation of value and are not restricted to utility. The question is what contexts are most relevant to what values?

At the very least, there is a loss of information when values are reduced to monetary terms. This often becomes evident when individual values are extrapolated to the population. The aggregate value reported by valuation studies is often sizeable compared with those agreed for other social priorities. The problem often arises, not from the question in principle, but from the practical characteristics of eliciting values by means of a survey. The brevity of the process and use of hypothetical scenarios makes it difficult for a respondent to adequately consider the limits on their ability to pay and the range of alternatives available. Individually, a person could indeed express a genuinely high maximum willingness-to-pay for an environmental good. Another person could express a similar high willingness-to-pay but do so without full consideration of the alternatives. There can also be much inconsistency between individuals' assumptions of the amounts they are expressing a value for. Large aggregate amounts may reveal that the process has failed to take account of the many factors. Although this failure is commonly a consequence of the procedure, rather than of valuation methods in general, these methods often fail to explore the range of social and cultural values and the related factors that restrict our flexibility of choice.

3.5 Deliberative approaches

In response to accusations of an undue emphasis on utility alone as captured by conventional economic valuation, there have been calls to develop a more transdisciplinary methodological framework (Daniel et al., 2012). This is especially relevant to the environment given that individual motives may not be the most desirable approach for valuing environmental goods that are often shared as public or common pool goods and which are vulnerable to over-exploitation (Vatn, 2009). Furthermore, individual approaches, especially when expressed in terms of aggregate willingness-to-pay are almost inevitably influenced by the income of those contributing the bids.

The Ecosystem Approach provides a route through which to examine alternatives or complementary approaches to a singular reliance on monetary valuation It is essentially a strategic framework of adaptive management of ecosystems that includes cross-sectoral integration of information and decision-making (Fish, 2011). It aims to understand multi-causality by disentangling the respective role of social, biophysical and socio-ecological influences. "Analytical deliberative decision-making" is an essential part of this process that takes into account diverse values, or value plurality, along with an understanding of the ecosystem's complexity and uncertainty (Fish, 2011).

Group deliberative methods are used for the discussion of alternative scenarios and values within small groups of people. The process permits the participants to gain a greater understanding of other people's point of view, including their value systems and their relationship with nature. It is also arguably more democratic than willingness-to-pay methods in that the respective income levels of the participants do not unduly influence the outcome. This approach potentially provides for equitable solutions.

Values are socially constructed over time (value construction) in contrast to the spontaneous statements made within survey-based economic valuation methods (Norton et al., 1998; O'Hara, 1996). The ecosystem, including the interplay between ecosystem processes and services, is complex. Therefore, a deliberative approach can be used to overcome bounded rationality by conveying the information that people need to arrive at a better informed decision. Social learning is a feature of this process. It can follow the careful provision of relevant technical or scientific information by the facilitator, but should equally enable the facilitator to obtain an insight into the participants' values, frames of reference, knowledge systems and their interaction with nature (Garmendia and Stagl, 2010; Gomez-Baggerthan and Martin-Lopez, 2014).

One form of the deliberative approach is the citizen jury (Kenyon et al., 2003). This can involve a series of workshops in which participants are presented with a range of information and asked to adjudicate in the interests of fellow citizens. The approach can provide a useful complement to the individual perspective taken by economic valuation. However, while participants are asked to take a citizen perspective, they themselves, along with the wider population who might be affected by the outcome, will hold a range of values associated with social and personal objectives. This can introduce a risk of role playing.

Any deliberative process must distinguish between the prior interests and values that the participants bring to the process (Vatn, 2009). This requires that the deliberation is active, i.e., that there is full interaction, exchange of information and debate. Used in this way, deliberation can be valuable for exploring the range of perceptions and values that participants have. These types of fora have also been used for deliberative monetary valuation to arrive at agreed willingness-to-pay values. The approach can be superior to individual expressions of willingness-to-pay as values may be agreed following a process of social learning and consideration of other people’s perspectives. There are limitations, though, in terms of the typically small numbers of people who can be involved in deliberation and the transferability of the values elicited to a larger population. Consequently, deliberation may be most useful within a multi-pronged approach or as a decision support method.

In a deliberation on the Inner Forth estuary landscape in Scotland, Kenter (2014) found that this process reduced willingness-to-pay amounts by 73% to levels he considered to be more representative of social than individual priorities. Furthermore, although participants still gave a high priority to biodiversity outcomes, the rated importance attached to economic development considerations increased during the series of meetings. The approach was combined with a questionnaire exploring the role of values-beliefs-norms in line with the New Economic Paradigm so as to identify the evolution of participants’ attitudes through the process (Schwartz, 1977). The study indicates that deliberative monetary valuation may be well-placed to handle trade-offs between competing social objectives. It suggests that choice experiments could be usefully employed in this context too with heterogeneity of preferences potentially informed further by the elicitation of information on value-norms-beliefs. In the final analysis, however, deliberative monetary valuation still assumes an ultimate commensurability of values, i.e., that values can be expressed in monetary terms.

Socio-ecological approaches can form part of the deliberative approach. These aim to take account of the relationship between people's behaviour and the environment. In many real world situations, examined for instance by Ostrom (2009), people's inter-relationship with the environment is determined through the social institutions that govern the use of common pool resources. An understanding of values and peoples' response to hypothetical scenarios cannot be understood without reference to this context. Another use of the socio-ecological approach is in the deliberative setting to make people more aware of the functioning of the environment, its inherent uncertainty and its possible fragility (Straton, 2006). This could be useful for involving stakeholders who have a direct impact on the management of an environment. Equally, it could be as useful in cases where people feel an attachment to the environment, for instrumental, ethical and aesthetic reasons, but are disenfranchised by having no such input into its management.

Lo (2011a) is more sceptical on the value of deliberation and argues that democratic deliberation does not necessarily lead to good decision making. He states that there is no guarantee of a successful outcome and that participants may remain attached to certain belief systems throughout the process. This is not an unfamiliar outcome. Complex or contentious issues often require long periods of time to resolve, especially where competing value systems or beliefs are present. Successful outcomes often do occur after a social learning process. However, most research-inspired deliberative exercises do not have sufficient resources or time to allow for the completion of such a process. If people are not strongly attached to particular elements, then resolutions might be possible. However, where our interest combines social and environmental values, then there is a strong prospect that the deliberative process will not be entirely satisfying to all participants. Furthermore, there is a risk of achieving a false consensus for the sake of demonstrating the merits of the method. Indeed, Lo highlights the risk of top-down influences whereby research teams deliberatively, or inadvertently, aim to secure successful outcome.

Lo argues instead that, given enough information, including information of competing values, most individuals are capable of internalising a range of positions to arrive at a more moderated valuation. However, this could be a rather optimistic outcome and one that omits the very useful group dynamics and social learning that can be provided by deliberative meetings comprised of a number of participants. A good deal has been written on the merits of well-moderated focus groups and workshops in providing participants with the opportunity to mentally articulate and debate positions. Important information on less mainstream values can be lost. Consensus positions are one outcome, but at least as useful at a practical level is the ability of the process to provide insights into the strength of people's values, even if they remain doggedly attached to a particular point of view (McDaniels et al., 1999) .

Ideally, a method is needed that can approximate a transdisciplinary approach. Baggerthan and Martin-Lopez (2014) argue that conventional deliberation amounts to no more than a hybrid approach. Deliberative monetary valuation aspires to be transdisciplinary, but falls short in this ambition because it ultimately needs to reduce values to a monetary/economic value context. Nevertheless, if the limits to the approach are accepted from the start and are communicated adequately to decision makers, deliberative monetary valuation can still be a useful means to explore competing values and to approach some level of agreement. On the basis of the Kenter study of the Inner Forth, it seems that this approach has merits even though it ultimately falls back on monetary willingness-to-pay values that arguably fail to achieve commensurability. A hybrid approach is one that accepts that values are often incommensurable, but that deliberation may still be complementary to economic valuation and useful for providing additional information through which to interpret different values and value contexts.

The alternative approach is to identify where there is strong and weak commensurability based on the familiar notions of strong and weak substitutability. Martinez-Alier et al (1998) argue that incommensurability does not have to imply incomparability and that, as a discipline, economics has always acknowledged plurality of values, for example between market based and socialist based allocations of resources. Their argument is that values that are incommensurable, but that these values can still be compared in a multi-criteria analysis (MCA). MCA is by nature multidimensional. It does not require that criteria have to be optimised at the same time, only that they can be compared so that a compromise solution can be identified.

3.6 Recommended valuation approach

On the basis of the literature, we arrive at the following observations for possible use in OPERAs.

1. The objective

Firstly, it is important to identify and remain aware of the objective of the exercise. Ecosystem services are an anthropocentric phenomenon in that the value of nature is recognised to the extent that it provides value to human beings. This value includes both instrumental values and non-instrumental values.

Ecosystem service assessment can be used to improve the management or protection of the environment for the benefit of human beings. Fortunately, the range of human values, combined with the need to sustain the flow of ecosystem services for our own well-being, means that this process typically highlights the benefits of maintaining and protecting fundamental ecosystem processes. Consequently, any integrated method of ecosystem service valuation should be practical or operational (Gomez-Baggerthan and Martin-Lopez, 2014). The particular approach will depend on the nature of the objective. This could include, but not be restricted to, setting priorities for policy, increasing awareness, providing information for development and spatial planning, understanding heterogeneity of preferences or to undertake environmental accounting. To date, however, the analysis of local socio-cultural values has tended towards research rather than operational interests (Ambrose-Oji and Pagella, 2012). Policy makers do not typically need philosophic debate on values. They need guidance on how to maximise economic and social welfare in a manner that is sustainable and does not unduly disadvantage the interests of particular social groups.

2. Commensurability of values

We have yet to find a method that can bring together incommensurate values in a convincing integrated valuation. However, we do have the tools that can be used to identify and compare values. These tools are not without their shortcomings and the development of improved or new tools is desirable. One means to compare the strength of people's attachment and valuation of ecosystem services is to place people in a position whereby they have to contemplate trade-offs between alternatives. The need to make trade-offs is not the exclusive preserve of economic decision-making. It is a familiar aspect of daily life.

Choice experiments have been used by economists and others to explore people's willingness to trade off one alternative for another. Using this tool, decision-makers can be informed of when, on the one hand, trade-offs are permissible and when, on the other hand, no trade-off can be contemplated. The latter position often reveals lexicographic preferences that could be linked to incommensurate values. The choice experiment design should be sufficiently informed by prior investigation to confirm this position, and to explore the limits and context to these preferences as well as to identify by whom they are held. If no trade-off can be made and the motivations appear genuine, then incommensurability may indeed be present.

A limitation of choice experiments is the requirement for goods to be broken down into a small number of attributes. Deliberative approaches frequently reveal that people mentally bundle goods together, especially when applying values to cultural ecosystem services (Bieling et al., 2014). Consequently, many environmental goods may be perceived and valued in a more holistic form. The reference to ecosystem services could be argued to be reductionalist too, but it does have the merit of identifying flows of services which may influence the supply and condition of a greater number of environmental goods or attributes. Some flows might be easier for participants in a deliberation process to distinguish than selected attributes of final goods or benefits..

A consideration in the use of choice experiments by economists is the inclusion of a monetary attribute. The monetary attribute usefully focuses attention, but can be difficult to represent in a meaningful way where public goods are concerned. Indeed, the monetary attribute can be the object of too much attention. It can cause preferences to be framed in an unduly utilitarian dimension or, alternatively, can provoke resistance on the basis of incommensurability of values. “Attribute non-attendance” may occur where respondents either place too much emphasis on the price attribute or, otherwise, treat it as being simply hypothetical and peripheral to the main choice (see xxx). Alternative approaches can be used from which monetary estimates can be derived indirectly, for example travel time. However, it can still be challenging to identify suitable payment vehicles that do not dominate the other attributes in the choice set.

3. Measuring values using deliberative methods

Although this chapter has acknowledged the limitations of a singular dependence on economic values and has explored the merits of deliberative approaches, economic welfare approaches have a trump card in providing measureable output, albeit linked to a particular theoretical foundation. Deliberative methods are evidently useful for raising people’s awareness of the issue and of the perceptions of other participants. However, pure deliberative approaches can be weak on providing a useful record of the process. This is one reason for the popularity of deliberative monetary valuation in which progressive changes in willingness-to-pay amounts are recorded as changing values during the course of the deliberation.

Ranking and scoring methods represent an alternative form of weighting to the output from trade-offs. However, the use of ranks and scores can rest on weak foundations that leaves the output vulnerable to inconsistency. However, scores can begin to converge during a series of deliberative exercises as participants respond to objective information or better understand others’ positions. While this convergence is not a requirement of understanding diverse values, it can have practical merit for achieving consensus on policy or decision-making. Germandia and Gamboa (2012) describe how methods can be applied to validate the “robustness of the analysis” through the use of a hierarchical clustering process. The clustering represents what they call a “compression process” whereby key services are identified. They show how the output can then be used to input into a participatory multi-criteria analysis of a more restricted number of alternatives (discussed below).

4. Bringing ecological criteria into the assessment - distinguishing between socio-cultural values and ecosystem services

If it is our intention to identify the role and value of ecosystem services from a socio-cultural perspective, then it is important to distinguish where these values relate to the ecosystem and where they do not Bieling (2012) describes a cultural valuation exercise in which the biotic factors only really featured in around two-thirds of the values that people expressed for the environment. Other examples of deliberation have demonstrated how valuation exercises have identified low values for regulating and provisioning services (Martin-Lopez et al., 2013) even where small group deliberation has permitted time for social learning. However, this result is often a consequence of how ecosystem services are presented. Participants may, for example, be informed of the importance of aquatic invertebrates for maintaining water quality, but it is asking much of participants to compare these invertebrates with more familiar species such as birds and mammals. Particular challenges emerge when discussing possible thresholds to the maintenance of these services.

The Maslow hierarchy, or the Human Scale Development Matrix, may provide a simple and accessible means to communicate the relevance of ecosystem services. Both can be used to identify, in the first instance, those services that are important to meet basic human needs, i.e., provisioning and certain regulating services, as well as other services that are important to higher level needs such as self-actualisation. The latter accords more closely to social-cultural values and therefore to cultural ecosystem services that contribute to end benefits.

A useful follow-up approach is to call upon group participants to assist with the development of a systems-type diagram that identifies the linkages, and the strength of linkages, between different ecosystem services and final benefits, for example between biodiversity and recreation, or between water quality and health. The approach is not new, but Kenter (2014) showed how this could be usefully combined with deliberative monetary valuation in the case study in Scotland. Scientific input is needed at this stage and should evidently be introduced with careful facilitation to avoid leading participants towards particular conclusions. This scientific input can, though, include relevant considerations with regard to the nature, sustainability and vulnerability of ecosystem services and the beneficial outputs to which they link.

Ultimately, we all depend on provisioning and key regulating services. Nevertheless, trade-offs can be made at local level, for example on the amount of land allocated to intensive agriculture or to biodiversity. For example, regulating ecosystem services are important for good water quality. This will be important for activities such as agriculture or fishing, but also for activities such as angling or wildlife viewing, as well as for various non-use benefits. These final goods and services fall into the cultural ecosystem service domain. The issue is less one of identifying the total value of a benefit, but rather of identifying the degree to which changes to the supporting, provisioning or regulating service can be permitted relative to other benefits that do not derive from the natural environment.

5. Multi-criteria analysis

Ultimately, values need to be brought together for the purpose of strategic environmental management. Scoring and weighting methods can be useful, especially where they are used to chart and record the deliberative process and the amount of social learning that has occurred. The clustering approach described above (Garmendia and Gamboa, 2012) is useful in this regard. Trade-off analysis can be employed in this final step applied to a particular environmental issue. Meaningful and well-balanced trade-offs can be incorporated into a choice experiment following the identification of the most relevant ecosystem services benefits in the clustering process.

Alternatively, MCA can be used as the final step to draw directly on the scores and weights provided previously. Lo (2011b) describes how MCA can handle different criteria in a process of “analytical deliberation” as opposed to democratic deliberation that is dependent purely on a discursive approach. However, there are benefits in combining the two approaches. MCA can be used to reveal where there is strong commensurability, weak commensurability and no commensurability. If there is a potential transdisciplinary overlap it will be found in the second of these categories. Where there is weak commensurability, there is an opportunity for comparability aided by practical judgement (Martinez-Alier et al., 1998; O'Neill, 1993). It is in this space where a pragmatic hierarchy of values can be drawn (Gregory et al., 1993) or where some compromise or trade-offs are possible.

Furthermore, Garmendia and Gamboa (ibid) describe how out-ranking MCA methods can be used to partially incorporate the non-compensatory positions that characterises strong commensurability. In this process, group participants are called upon to apply scores to certain criteria. The scores are analysed through a clustering process (as discussed above) which reduces the alternative to a tractable number prior to a further round of deliberation. At this stage, those criteria that are dominated are omitted from a pair-wise comparison. The resulting outranking matrix of preferred alternatives provides the basis for a final MCA. The procedure provides a means to include trade-offs within the deliberative process, but also to record and validate this process without recourse to monetary estimates or the exclusion of all incommensurable values.

3.7 Conclusion

Non-market economic valuation, deliberative socio-cultural valuation, and deliberative monetary valuation all share a common objective. All of these methods can be used to provide an insight into what is it that maximises social welfare or well-being. Economic valuation methods can be criticised for capturing only a single dimension of value, namely that associated with individual utility. In principle, economists argue that ethical or social considerations also enter the utility function. However, this claim is difficult to support with environmental goods, i.e., where these are public goods, where there is a shared dependence on the environment, or where there is an ethical argument for equal access to environmental goods.

Deliberative settings have the capacity to inform people about complex ecosystem services and about other people's values and relationship with the environment. Deliberative monetary valuation has been used to strengthen the output from conventional economic valuation tools. Nevertheless, significant problems remain in addressing the challenge of incommensurability of values. No transdisciplinary tool yet exists that can adequately represent the range of values that exist. However, this chapter has discussed the key considerations and has introduced examples of promising approaches that can be applied to gauge the extent of true incommensurability and to meet the practical needs of ecosystem management.

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4 Developments in valuation and valuation methods

In this chapter we present the results of several studies aimed at developing economic valuation and economic valuation methods. The topics addressed in the following four subsections are:

- Reference point dependence in monetary valuation;
- Effects of ignoring the payment vehicle in choice experiments on value estimates;
- Effects of including a price attribute on choices and trade-offs in a choice experiment;
- Regret minimisation models as an alternative to utility maximisation models.

4.1 Reference dependence and the WTA-WTP disparity[8]

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4.1.1 Introduction

Stated preference methods such as contingent valuation and choice experiments are not often used in flood risk valuation studies (Birol et al., 2009;Brouwer et al., 2009; Dekker, 2012), but there is an emerging stated preference literature on demand for flood insurance, both in developed and developing countries (Botzen and van den Bergh, 2009; Brouwer and Akter, 2010; Botzen and van den Bergh, 2012). These studies typically investigate household willingness to pay (WTP) for flood insurance under different flood probabilities and damage covers. The general expectation underlying the willingness to pay for insurance is that people are risk averse when their decision involves potential losses under low probability-high impact conditions, and corresponding individual choice behaviour is motivated by a desire for security. Low probability and high impact conditions refer to situations in which the occurrence of an event is low, but the effect of the event may cause large losses.

Existing studies in the context of flood insurance demonstrate that WTP depends on both objective and subjective risk measures (e.g., Brouwer and Schaafsma, 2012). The study by Brouwer and Schaafsma (2012) is so far the only study that also examines willingness to accept (WTA) compensation in the context of increasing flood probabilities due to climate change, related to controlled flooding in designated flood disaster zones. In this study we are interested in both WTP and WTA values for changes in flood probability, and conduct choice experiments to distinguish between the two measures under different flood probability reference points. The empirical literature shows that there exist substantial differences between WTP and WTA values, motivated by endowment effects and loss aversion. Prospect theory states that people are more sensitive to losses than to gains given their reference points, and that changes in utility are dependent on the reference point chosen (Kahnemann and Tversky, 1979). Although several empirical studies have confirmed the patterns predicted by this theory, empirical evidence for these behavioural patterns in the specific context of catastrophic flood risks is to our knowledge absent. Further testing of this

[8] The most recent and full version of this section has recently appeared as: Koetse MJ, Brouwer R, in press, Reference Dependence Effects on WTA and WTP Value Functions and their Disparity, *Environmental and Resource Economics*, doi: 10.1007/s10640-015-9920-2.

issue is required to examine how alternative utility function specifications affect individual household risk perception and behaviour.

In order to test the implications of prospect theory in our choice experiment, we vary flood probabilities in the status quo in both a WTP and a WTA experiment, and use separate experiments for each status quo situation, in order to test for reference dependency and its impact on the WTP-WTA discrepancy. Our main objective is to compare and test equivalence of two different economic measures for the same welfare loss, measured through WTP to prevent the welfare loss (equivalent variation) and WTA compensation for the welfare loss (compensating variation). Our central hypotheses are: (1) due to loss aversion and endowment effects there is a gap between WTP and WTA, (2) WTP and WTA estimates for a change in flood probability are dependent on the reference flood probability level, and (3) the WTP-WTA gap increases as we increase the reference flood probability level. To this end we conduct four separate choice experiments using four split samples with similar socio-demographic groups living in the same case study area, i.e., a WTP experiment with two status quo reference situations, and a WTA experiment with two status quo reference situations. For each experiment the relative flood probabilities are kept constant in order to be able to test the effect of reference point dependency.

4.1.2 Case study background and choice experiment design

The IJsselmeer is one of the largest freshwater buffers in Europe. This buffer is used primarily during the summer season as one of the main sources of water supply for agriculture and residential household water demand in The Netherlands. Until 2050 a shortage in water supply may result as a consequence of climate change and socio-economic developments. For these reasons the Dutch government is considering a future increase in the IJsselmeer water level. A small increase in the water level (30 cm) during the summer season already substantially increases the freshwater buffer, and requires relatively limited investments. However, in the longer run after the year 2050, it is likely that more substantial water level increases are necessary. Increases in the water level of more than 30 cm has a number of consequences in and around the IJsselmeer. Without additional government investments in flood control flood probabilities will increase, wildlife habitat along the shores of the IJsselmeer will disappear, and bird populations will decrease in size and variation. Research into the costs and benefits of measures to mitigate the associated negative effects of increases in water levels shows that raising IJsselmeer water levels along with sea level rise will result in substantial costs (Bos et al., 2012).

In this study we assess preferences of people who live in the vicinity of the IJsselmeer for potential future changes in and around the IJsselmeer, and to translate these preferences into monetary values. We distinguish between WTP and WTA for changes in the expected impacts of future policy scenarios in the IJsselmeer study area related to increasing water levels, i.e., flood probability, wildlife habitat and bird populations, by conducting separate WTP and WTA choice experiments. We also vary the reference points for flood probabilities in the WTP and WTA experiments in order to assess how this impacts the WTP and WTA estimates and the WTP-WTA discrepancy. To this end, four versions of the same choice experiment were created: WTP1, WTA1, WTP2, WTA2. By keeping attributes and attribute levels constant across the four versions, but by changing the status quo, we can compare WTP and WTA for identical changes. By varying attribute levels for flood probability between experiments 1 and 2 the impact of reference points on the WTP-WTA gap can be tested. The attributes and their levels for the four experiments are summarised in Table 2.

Table 2. Attributes and their levels in the four choice experiments

Attributes	Attribute levels
Raising the dikes	▪ No (baseline scenario in WTP version) ▪ Yes (baseline scenario in WTA version)
Flood probability WTP1 & WTA1	▪ Once every 10,000 years (baseline scenario WTA1) ▪ Once every 2,000 years ▪ Once every 1,000 years (baseline scenario WTP1)
Flood probability WTP2 & WTA2	▪ Once every 5,000 years (baseline scenario WTA2) ▪ Once every 1,000 years ▪ Once every 500 years (baseline scenario WTP2)
Bird population	▪ 0 % (baseline scenario WTA) ▪ –10 % ▪ –30 % (baseline scenario WTP)
Type of shore	▪ No additional shores (baseline scenario WTP) ▪ Additional shores not connected to the dike ▪ Additional shores connected to the dike (baseline scenario WTA)
Change in annual local tax (increases in WTP, decreases in WTA)	▪ €0 (baseline scenario) ▪ €60 ▪ €100 ▪ €180

In order to test prospect theory and reference point dependency, we vary the flood probabilities in the status quo in both the WTP and WTA experiments, and use separate experiments for each status quo situation. Four separate choice experiments are designed, i.e., a WTP experiment with two status quo situations or baseline scenarios and a WTA experiment with two baseline scenarios. In each experiment the change in relative flood probabilities is kept constant. The levels in the first set of WTP/WTA experiments (WTP1 and WTA1) are once every 10,000 years, once every 2,000 years and once every 1,000 years. The levels in the second set of WTP/WTA experiments (WTP2 and WTA2) are once every 5,000 years, once every 1,000 years and once every 500 years.

We generate a fractional factorial design, using the restriction that policy scenarios should include an improvement (in the WTP version) or deterioration (in the WTA version) of at least one of the three non-monetary attributes. We generated a statistical design consisting of 15 survey versions with 10 choice tasks each, and each respondent was randomly assigned to one of these 15 versions. In order to ensure a symmetrical design for the WTP and WTA versions, the local tax levels in the two policy scenarios in the WTA versions were switched compared to the WTP version. Although we use the same design in the WTP and WTA experiments, an increase of say 180 Euro in the WTP version in alternative 1 vis-à-vis an increase of say a 100 Euro in alternative 2 implies that alternative 2 is financially more attractive by 80 Euro. If the same design would be used for the WTA version, the signs of the tax change are switched, implying that alternative 1 is now financially more attractive by 80 Euro. In order to avoid this asymmetry, the tax levels for the two policy scenarios in the WTA version are switched compared to the WTP version.

In the choice experiment both the willingness to accept negative effects and the willingness to pay to avoid negative effects are analysed, and the status quo or baseline scenario is changed accordingly. The baseline scenario in the WTP experiment is a situation that is most unfavourable with respect to flood probability, shores in the IJsselmeer and impacts on bird populations, but in which no tax increases take place. The two alternative policy scenarios in the WTP version are more favourable with respect to at least one of these three non-monetary attributes, accompanied by an increase in annual local taxes. The baseline scenario in the WTA version is a situation that is most favourable with respect to flood probability, shores in the IJsselmeer and impacts on bird populations, but in which there are no tax reductions. The two alternative policy scenarios in the WTA version are less favourable with respect to at least one of the three non-monetary attributes, and are accompanied by a compensation in the form of a decrease in annual local taxes. Examples of WTP and WTA choice tasks are presented in Figures 4 and 5.

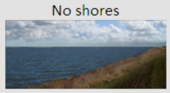
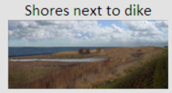
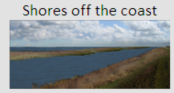
	OPTION 1 **No increase in dike height**	**OPTION 2** **Increase in dike height**	**OPTION 3** **Increase in dike height**
Flood probability	Once every 1,000 years (3% in 30 years)	Once every 1,000 years (3% in 30 years)	Once every 10,000 years (0.3% in 30 years)
Type of shore	No shores 	Shores next to dike 	Shores off the coast 
Bird population	Decrease of 30%	No change	Decrease of 10%
Annual local tax	No change	60 Euro per year MORE	100 Euro per year MORE
You prefer:	○	○	○

Figure 4. Example choice card experiment WTP 1

	OPTION 1 Increase in dike height	OPTION 2 No increase in dike height	OPTION 3 No increase in dike height
Flood probability	Once every 10,000 years (0.3% in 30 years)	Once every 2,000 years (1.5% in 30 years)	Once every 1,000 years (3% in 30 years)
Type of shore	Shores next to dike 	No shores 	Shores off the coast 
Bird population	No change	Decrease of 30%	Decrease of 10%
Annual local taxng	No change	60 Euro per year LESS	100 Euro per year LESS
You prefer:			

Figure 5. Example choice card experiment WTA 1

For data collection we used a Dutch internet panel owned by TNS-NIPO, which includes more than 200,000 households from The Netherlands. In the TNS-NIPO panel around 6,800 respondents live in the postal code areas around the IJsselmeer. From this set four independent samples were drawn, one for each of the four choice experiment versions. A representative sampling procedure was employed based on age, gender, household size, social class (education and profession) and residential location, using the socio-demographic composition of the TNS respondent sample in our population area as the reference situation. For each of the four choice experiments 375 households were invited to complete the survey. The total number of respondents was 1,208, more or less equally divided across the 4 subsamples (n=297, 298, 299 and 314) and the postal code areas around the IJsselmeer, yielding a response rate of around 80 percent. Comparing the composition of the population with the compositions of the four samples reveals that that these are very. Any differences observed in the estimation results presented in the next section between the four experiments are therefore not expected to be caused by differences in respondent socio-demographic background and household composition.

4.1.3 Statistical model and estimation results

Preferences are modelled in terms of McFadden's (1974) Random Utility Model (RUM). To account for preference heterogeneity we estimate a Mixed Logit (ML) model, in which the preference parameters for the non-price attributes are allowed to vary across respondents. Recent applications of ML-models have shown that this model is superior to the standard multinomial logit model in terms of overall fit and accuracy of welfare estimates (e.g., Breffle and Morey, 2000; Layton and Brown, 2000; Morey and Rossmann, 2003; Provencher and Bishop, 2004; Brouwer et al., 2010). Mixed logit models account for respondent differences (preference heterogeneity) and repeated choices (Train, 2003).

The monetary attribute included in the choice experiment implies welfare estimates can be derived (e.g., Hensher et al., 2005). The welfare measure represents the monetary value arising from a change in the bundle of policy scenarios, also referred to as the consumer surplus (CS). In our study economic welfare implications are estimated for the four different versions of the choice experiment, representing *equivalent surplus* in the case of the two WTP versions (willingness to pay for preventing a welfare loss) and *compensating surplus* in the case of the two WTA versions (willingness to accept the same welfare loss).

With respect to model specification we include the attribute levels of flood probability, type of shores and bird population as dummy variables. The necessary reference categories for the dummy variables in the WTP models are the attribute levels used in the WTP status quo, while in the WTA models the reference categories are the attribute levels used in the WTA status quo. We also include a constant in the status quo utility function in order to test, whilst controlling for changes in flood probabilities, whether increasing dike height has an effect on safety perception. In all models annual local tax is included as a continuous variable. We estimate a Mixed Logit (ML) model using 500 Halton draws from a uniform distribution. Estimation results for experiments WTP1 and WTP2, along with the WTP estimates and their standard errors, are presented in Table 3.

The results show that the preference heterogeneity, represented by the estimated standard deviations of the random parameter distributions, is generally large. Especially preferences for increasing the height of dikes, reducing flood probabilities to once every 5,000 and 10,000 years, creating shores next to the dike and preserving the entire bird population, display strong variation. All estimated means of the parameter distributions have the expected signs and are statistically significant at a critical significance level of at least 5%. Focusing on flood probabilities, the results show that WTP estimates for flood probabilities that are two and ten times smaller than in the reference category are very similar for the two experiments.[9] Interesting is that in both experiments the willingness to pay for flood probabilities ten times smaller is only slightly larger than the willingness to pay for flood probabilities two times smaller, suggesting an increasing *marginal* willingness to pay for a reduction in flood probability when flood probabilities increase.

[9] Two pretests were done in which we, among other things, tested whether respondents understood the flood probabilities. Although results of the pretests suggest that people have a good understanding of the probabilities represent.

Table 3. Mixed logit estimation results for the two WTP experiments

Attributes	WTP 1 b	t-value	wtp	se(wtp)	WTP 2 b	t-value	wtp	se(wtp)
Mean of random parameters								
Status quo constant								
No increase in dike height	–1.266	–3.48	–€ 60	€ 17.0	–1.754	–5.45	–€ 92	€ 16.6
Flood probability								
1 in 1,000 → 1 in 2,000	0.716	7.54	€ 34	€ 4.4	--	--	--	--
1 in 1,000 → 1 in 10,000	0.986	7.92	€ 47	€ 5.8	--	--	--	--
1 in 500 → 1 in 1,000	--	--	--	--	0.731	8.52	€ 38	€ 4.4
1 in 500 → 1 in 5,000	--	--	--	--	0.838	7.88	€ 44	€ 5.4
Type of shore								
No → Off the coast	0.438	4.69	€ 21	€ 4.4	0.379	4.40	€ 20	€ 4.4
No → Next to dike	0.568	4.98	€ 27	€ 5.2	0.522	4.94	€ 27	€ 5.3
Bird population								
–30% → –10%	0.967	9.43	€ 46	€ 4.7	0.663	7.63	€ 35	€ 4.4
–30% → 0%	1.113	7.98	€ 53	€ 6.2	0.884	8.20	€ 46	€ 5.4
Standard deviations of random parameters								
Status quo constant								
No increase in dike height	4.700	12.1	--	--	4.544	12.5	--	--
Flood probability								
1 in 1,000 → 1 in 2,000	0.131	0.64	--	--	--	--	--	--
1 in 1,000 → 1 in 10,000	1.149	7.91	--	--	--	--	--	--
1 in 500 → 1 in 1,000	--	--	--	--	0.170	0.55	--	--
1 in 500 → 1 in 5,000	--	--	--	--	0.899	6.92	--	--
Type of shore								
No → Off the coast	0.246	1.05	--	--	0.359	1.79	--	--
No → Next to dike	0.868	6.23	--	--	0.907	6.97	--	--
Bird population								
–30% → –10%	0.551	3.12	--	--	0.164	0.76	--	--
–30% → 0%	1.364	9.11	--	--	0.913	7.48	--	--
Fixed parameters								
Annual local tax								
Increase	–0.021	–17.2	--	--	–0.019	–18.1	--	--
Number of observations	2,970				2,980			
Log–L	–2,021				–2,094			
Pseudo R^2 (adjusted)	0.379				0.359			

Estimation results for experiments WTA1 and WTA2 are presented in Table 4. Although there are small differences between the WTP and WTA results, the general patterns are very similar. The substantial preference heterogeneity in the WTP models is also found in the WTA models, and increasing the height of dikes is again valued positively. An increase in flood probabilities strongly decreases utility and we again find are strong non-linear effects. The most striking difference between the WTP and WTA models is that estimated WTA values are substantially higher than estimated WTP values. In the next section we discuss this issue in more detail.

Table 4. Mixed logit estimation results for the two WTA experiments

Attributes	WTA 1 b	t	wta	se(wta)	WTA 2 b	t	wta	se(wta)
Mean of random parameters								
Status quo constant								
Increase in dike height	1.442	4.81	€ 208	€ 43.6	0.986	3.97	€ 181	€ 45.2
Flood probability								
1 in 10,000 → 1 in 2,000	–1.194	–8.00	–€ 172	€ 31.2	--	--	--	--
1 in 10,000 → 1 in 1,000	–1.620	–9.86	–€ 233	€ 37.4	--	--	--	--
1 in 5,000 → 1 in 1,000	--	--	--	--	–1.474	–8.96	–€ 270	€ 54.0
1 in 5,000 → 1 in 500	--	--	--	--	–2.015	–10.9	–€ 369	€ 68.1
Type of shore								
Next to dike → Off the coast	0.041	0.31	€ 6	€ 18.9	–0.347	–2.75	–€ 64	€ 25.4
Next to dike → No	–0.473	–3.19	–€ 68	€ 21.9	–0.977	–5.63	–€ 179	€ 39.9
Bird population								
0% → –10%	–1.090	–7.48	–€ 157	€ 27.4	–0.874	–6.65	–€ 160	€ 33.5
0% → –30%	–2.382	–10.7	–€ 343	€ 51.2	–1.712	–10.2	–€ 314	€ 56.8
Standard deviations of random parameters								
Status quo constant								
Increase in dike height	3.575	13.7	--	--	3.559	12.9	--	--
Flood probability								
1 in 10,000 → 1 in 2,000	0.972	4.41	--	--	--	--	--	--
1 in 10,000 → 1 in 1,000	1.009	5.23	--	--	--	--	--	--
1 in 5,000 → 1 in 1,000	--	--	--	--	1.257	6.11	--	--
1 in 5,000 → 1 in 500	--	--	--	--	1.392	7.77	--	--
Type of shore								
Next to dike → Off the coast	0.752	4.15	--	--	0.664	3.83	--	--
Next to dike → No	0.942	5.28	--	--	1.317	7.90	--	--
Bird population								
0% → –10%	0.946	5.07	--	--	0.683	4.16	--	--
0% → –30%	1.715	7.50	--	--	1.353	7.66	--	--
Fixed parameters								
Annual local tax								
Decrease	0.007	7.13	--	--	0.005	5.73	--	--
Number of observations	2,990				3,140			
Log–L	–1,763				–1,877			
Pseudo R^2 (adjusted)	0.462				0.455			

4.1.4 Reference dependence and the WTA-WTP disparity

In this section we test our first central hypothesis that due to loss aversion and endowment effects there is a gap between WTP and WTA for identical welfare changes. WTP and WTA estimates for changes in bird population are presented in Figure 6; WTP (WTA) estimates indicate the willingness to pay (required compensation) for preventing (accepting) a reduction of 30%. The two WTP curves are very comparable, and so are the two WTA curves. Both the WTP and WTA models show non-linear effects, but where there is decreasing marginal willingness to pay when the reduction in bird population gets smaller, there is an increasing marginal willingness to accept. The required compensation is four times larger than the related willingness to pay for decreases of

10%, and seven times larger for decreases of 30%. Stated differently, the WTP-WTA gap increases when welfare changes increase.

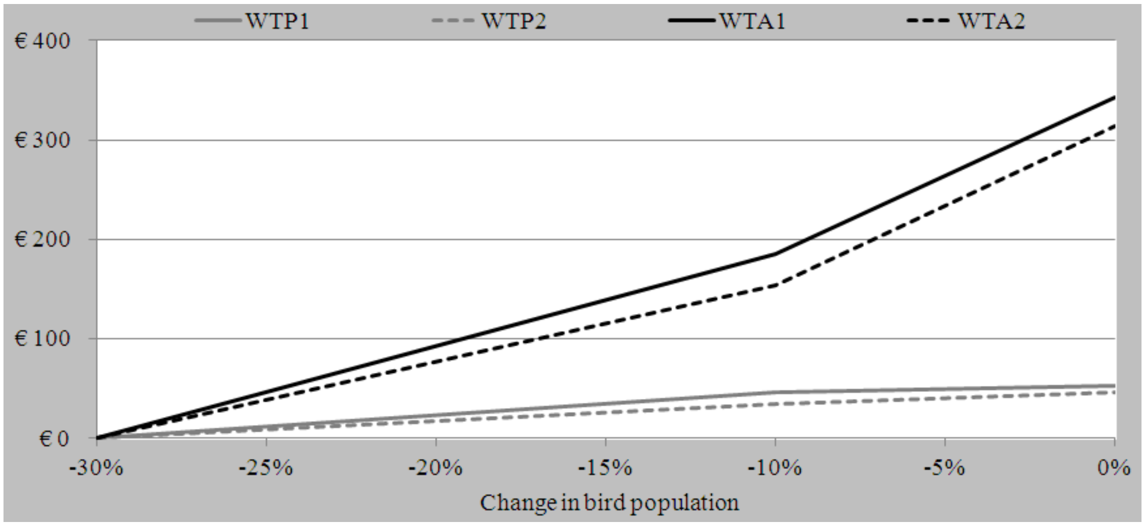


Figure 6. WTP/WTA estimates for preventing/accepting a decrease in bird population of 30% (in Euro per year)

In Figure 7 we present WTP/WTA estimates for preventing/accepting an increase in flood probabilities for experiments WTP1 and WTA1. The WTP (WTA) estimates indicate the willingness to pay (required compensation) for preventing (accepting) an increase in flood probability to once every 1,000 years. The WTP and WTA experiment show a similar non-linear pattern, with decreasing marginal WTP (WTA) estimates for decreasing flood probabilities. Interesting is that, similar to the situation for bird population, the difference between WTP and WTA estimates increases considerably when welfare changes increase. The required compensation is almost two times larger than the willingness to pay for a flood probability of once every 2,000 years, while it nearly five times larger for a flood probability of once every 10,000 years.

In Figure 8 we present WTP (WTA) estimates for preventing (accepting) an increase in flood probability for experiments WTP2 and WTA2. The WTP (WTA) estimates indicate the willingness to pay (required compensation) for an increase in flood probability to once every 500 years. The patterns are very similar to those in Figure 7; there are strong non-linear effects with decreasing marginal WTP and WTA when the flood probability decreases, and the difference between WTP and WTA increases for increasing welfare changes. The required compensation is almost 2.5 times larger than the willingness to pay for a flood probability of once every 1,000 years, while it nearly eight times larger for a flood probability of once every 5,000 years.

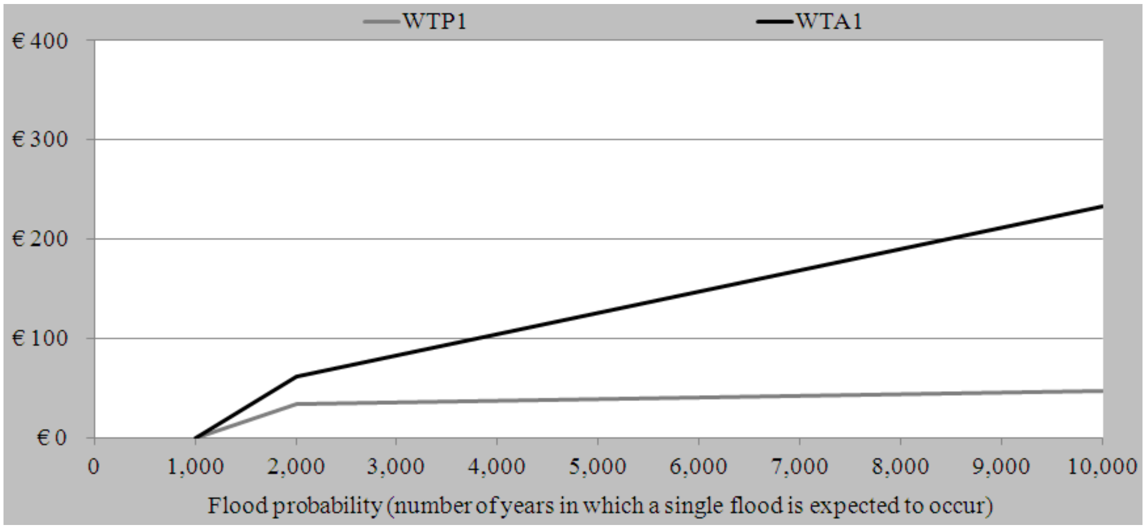


Figure 7. WTP/WTA estimates for preventing/accepting an increase in flood probability to once every 1,000 years (in Euro per year); experiments WTP 1 and WTA 1

In general we can conclude that, depending on the specific attribute and the specific attribute level, the WTA values from our experiment are two to eight times larger than the related WTP values. Moreover, both for bird population and for flood probability we find a strong increase in WTP-WTA differences when welfare changes increase.

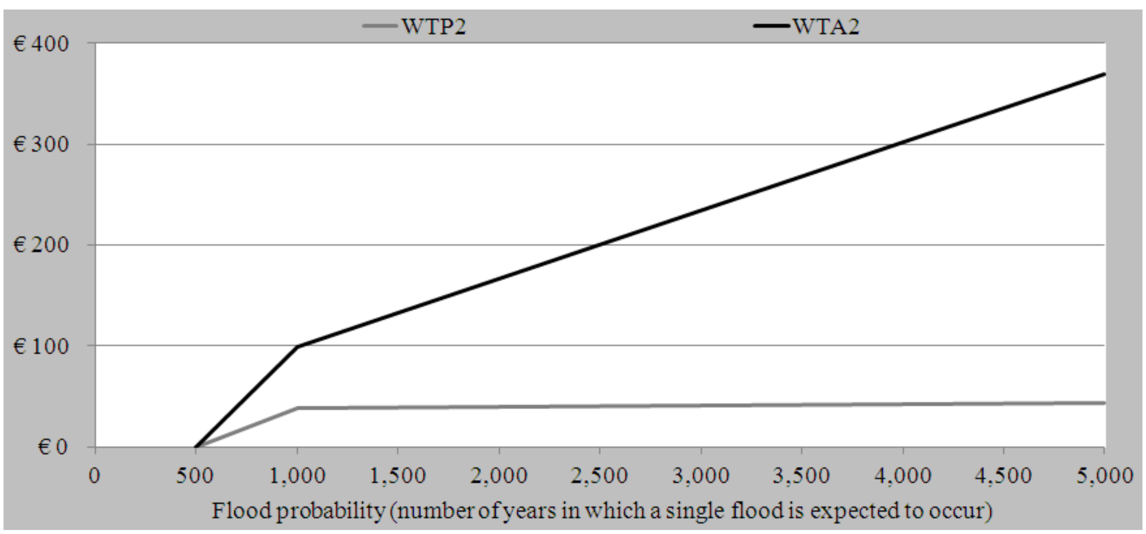


Figure 8. WTP/WTA estimates for preventing/accepting an increase in flood probability to once every 500 years (in Euro per year); experiments WTP 2 and WTA 2

4.1.5 Reference point dependence in flood valuation

Our second and third hypotheses are that WTP and WTA estimates for a change in flood probability are dependent on the reference flood probability level, and that the WTP-WTA gap increases as we increase the reference flood probability level. To test the second hypothesis we compare results from WTP1 with those from WTP2, and results from WTA1 with those from WTA2. The third hypothesis is tested by comparing the WTP-WTA difference in experiment 1 with the WTP-WTA difference in experiment 2. We start by making the WTP curves of experiment 1 and 2 more comparable. The reference point in WTP1 is a flood probability of once every 1,000 years, and by definition has a WTP value of 0. We therefore augment the WTP curve from WTP1 with the WTP for once every 1,000 years from experiment 2, making the reference point for both experiments equal to once every 500 years. We furthermore extrapolate the WTP value for once every 10,000 years in experiment 2 by using the WTP growth rate from once every 2,000 years to once every 5,000 years. The results are presented in Figure 9. The figure clearly shows that although the WTP *patterns* are very comparable, the WTP *values* for specific flood probabilities can be very different. Especially striking is that using a higher flood probability as a reference in WTP2 eventually leads to lower WTP values. To be specific, without reference point dependence one would expect the WTP for a flood probability of once every 5,000 years in WTP2 to be at least as large as the WTP for a flood probability of once every 2,000 years in WTP1. This is certainly not the case. The difference in WTP for a flood probability of once every 5,000 years between experiment 1 and 2 is around 30 Euro per household per year. Clearly the chosen reference point has an impact on the WTP values obtained. However, where the effect appears to be substantial at first sight, it turns out to be relatively small when compared to the WTA situation.

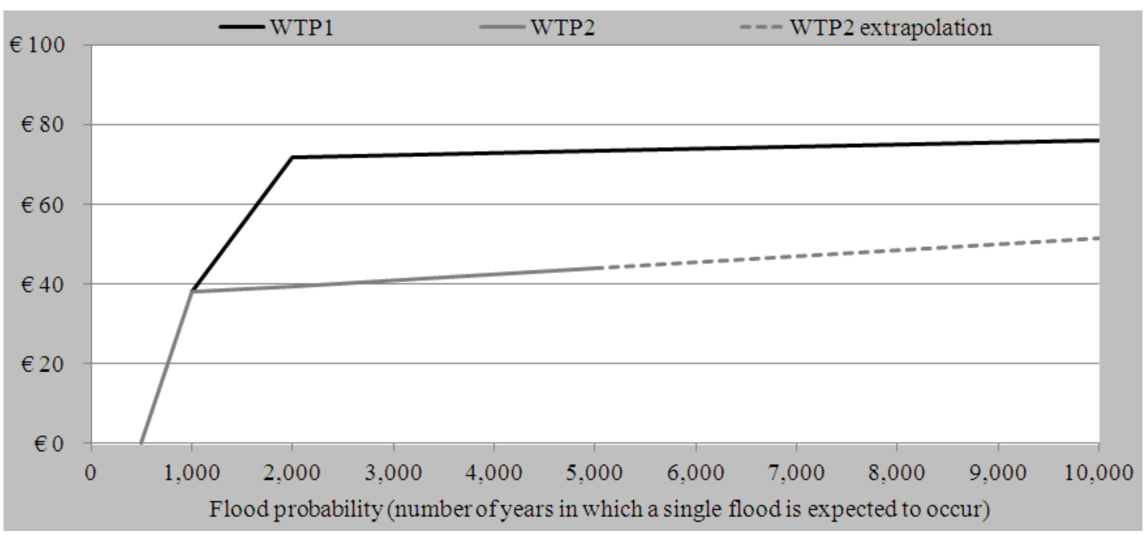


Figure 9. WTP estimates for preventing an increase in flood probability to once every 500 years (in Euro per year)

In order to make the two WTA curves comparable we carry out the same procedures as in the WTP situation. We did two extrapolations for WTA2; one based on the WTA growth rate from WTA1 (point B) and one based on the WTA growth rate from WTA2 (point A). The results are presented in Figure 10. The most important difference between the WTP and the WTA situation is that the differences in the WTA situation are larger, i.e., the impact of the chosen reference point on monetary welfare estimates appear to be larger for WTA than for WTP.

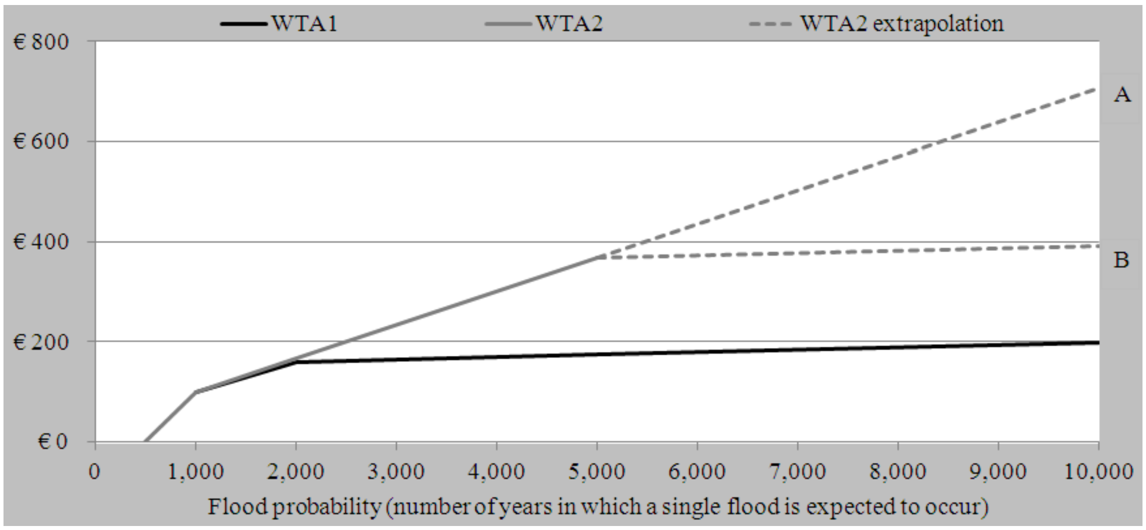


Figure 10. WTA estimates for accepting an increase in flood probability to once every 500 years (in Euro per year)

In order to illustrate the consequences of this finding more clearly we combine the WTP and WTA results in Figure 11. First, the figure shows that, as discussed earlier, WTP and WTA differences increase when welfare changes increase. Second, the WTP-WTA discrepancy is much larger for experiment 2 than for experiment 1, showing that reference points can have a substantial impact on the differences between willingness to pay and required compensation. To be more specific, for a flood probability of once every 5,000 years the WTP-WTA discrepancy is in experiment 2 is more than three times larger than in experiment 1. When for experiment 2 we extrapolate the WTA based on results from experiment 2 itself (WTA2a) the WTP-WTA difference for a flood probability of once every 10,000 years is more than five times larger in experiment 1 one than in experiment 2. When we extrapolate the WTA based on results from experiment 1, which may well be more plausible, the WTP-WTA difference for a flood probability of once every 10,000 years is still more than three times larger in experiment 1 one than in experiment 2.

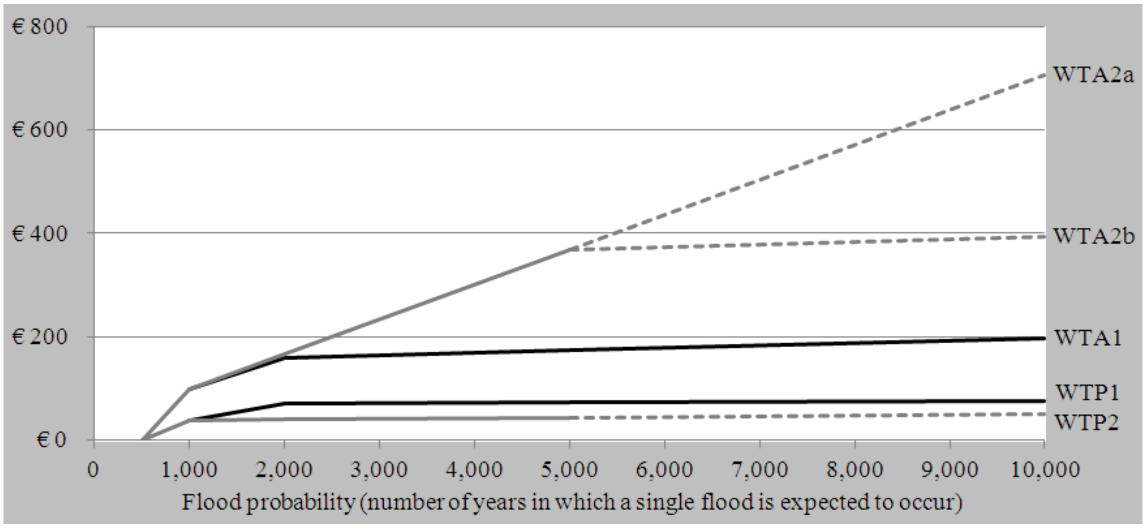


Figure 11. WTP/WTA estimates for preventing/accepting an increase in flood probability to once every 500 years (in Euro per year); dotted lines are extrapolations

4.1.6 Conclusions and discussion

The bulk of the empirical evidence shows that there are large differences between estimates of willingness to accept and willingness to pay for identical changes in welfare. This is at odds with expected utility theory, which assumes that, after controlling for income effects, equal welfare effects of gains and losses. As an alternative decision making theory under risk and uncertainty, prospect theory predicts that people are more sensitive to losses than to gains given their reference points, and that changes in utility are dependent on the reference point chosen. In this study we test these issues in a low probability high impact context, using the IJsselmeer area in The Netherlands as a case study.

Our main objective is to compare and test equivalence of two different economic measures for the same welfare loss, measured through WTP to prevent the welfare loss (equivalent surplus) and WTA compensation for the welfare loss (compensating surplus). Our central hypotheses are that due to loss aversion and endowment effects there is a gap between WTP and WTA, and that this gap increases as we increase the reference flood probability level in the choice experiment. To this end we conduct four separate choice experiments using four split samples with similar socio-demographic groups living in the same case study area, i.e., a WTP experiment with two status quo reference situations, and a WTA experiment with two status quo reference situations.

Our first central hypothesis in this paper was that there are differences between WTP and WTA for identical welfare changes. These differences turn out to be large, and depending on the specific welfare change the WTA values are two to eight times larger than the associated WTP values.

Moreover, both for changes in bird population and for changes in flood probability we find that the difference between WTP and WTA grows when welfare changes are larger. Our second central hypothesis was the WTP-WTA gap increases as we increase the reference flood probability level in the choice experiment. Our findings confirm this hypothesis. We show that, apart from the impact of the magnitude of welfare changes on WTP-WTA differences, the WTP-WTA discrepancy in flood valuation is considerably larger in those choice experiments that contain higher flood probabilities as a reference. More specifically, for a flood probability of once every 5,000 years the WTP-WTA discrepancy is three times larger for a reference situation of once every 500 years than for a reference situation of once every 1,000 years. Both findings confirm important features of prospect theory, and show that welfare changes related to changes in low probability high impact situations strongly depend on the current situation (status quo) and on the magnitude and direction of change.

Our findings have several implications, especially in the field of stated preference research for the valuation of environmental change. First, reference dependency implies that there may be strong temporal and spatial dynamics in the welfare consequences of environmental changes. More specifically, welfare consequences of similar environmental changes may vary over time because reference points change, and may differ between countries and regions because reference points are different. In order to reliably assess the welfare consequences of specific environmental changes, time- and space-specific research is therefore required. A second and related point is that transferring values obtained from previous research, for example values obtained from meta-analysis, to other regions and time periods may lead to large transfer errors if one does not account for the risk context and baseline risk levels in which existing values were elicited. Although this conclusion in itself is not new, and is supported by ample empirical evidence (see also Dekker et al., 2011), our findings suggest reasons for why these transfer errors may be as large as they are. Finally, given the pervasive disparity between WTA and WTP, choosing the appropriate welfare measure to assess the economic consequences of changes in public good provision is essential. The use of WTP as the most widely endorsed indicator of welfare change may underestimate the economic value of welfare changes when it comes to accepting losses and may lead to suboptimal design of environmental policies (see also Knetsch, 2010). However, our results show that WTA is more sensitive than WTP to both the scale of change that is being studied and the reference value. This implies that WTA values which are obtained from studies that assess a different range of possible changes, and that use a different reference value than is the case for the specific welfare analysis, may overestimate a welfare change. These observations call for case-study specific research on the welfare effects associated with changes in the quantity and/or quality of public environmental good provision. If value transfer is used instead, incorporating reference point effects in transferring values or functions is strongly advisable.

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4.2 Effects of tax non-attendance in choice experiments on value estimates and the WTA-WTP disparity[10]

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4.2.1 Introduction

Differences between willingness to accept (WTA) and willingness to pay (WTP) estimates continue to differ widely. These differences were first observed in hypothetical questions involving public goods (see Cummings et al., 1986), and many empirical studies have reported large WTA-WTP disparities since then. In a meta-analysis of WTA-WTP ratios Horowitz and McConnell (2002) show that large WTA-WTP disparities are widely spread, and moreover that they are prevalent across study subjects and study design, providing evidence that WTA-WTP differences are not just experimental artefacts. An especially worrying result in the context of valuation of public goods, which is often done using hypothetical non-market valuation methods, is that the WTA-WTP ratio is substantially higher for hypothetical experiments than for real experiments (see also List, 2003), and for public non-market goods than for private market goods.

Broadly three different types of explanations have been brought forward to explain the observed disparities between WTA and WTP (see also Mansfield, 1999). First, neoclassical explanations are the existence of income and substitution effects (e.g., Hanemann, 1991). Income effects arise due to the fact that increases and decreases in a certain monetary attribute have different consequences for a person's total income, which may affect their relative magnitudes. For public good provision substitution effects arise when people perceive that private goods in their choice set are an imperfect substitute for the public good that is being studied. The WTA for a loss in public good provision may then be higher than the WTP to obtain a gain. However, studies by among others Sudgen (1999) and Horowitz and McConnell (2003) show that disparities observed in reality cannot be reasonably explained by these income and substitution effects only. Another potential explanation within the neoclassical paradigm is provided by Zhao and Kling (2001), who argue that a WTA-WTP disparity may arise when consumers are uncertain about the value of a public good. Along the lines of real options theory, when consumes are asked to make a choice under uncertainty, and assuming significant costs of reversing the decision, they must give up the option

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to gather more information. Consumers have to be compensated for giving up this option value, implying a decrease in WTP and an increase in WTA, giving rise to a WTA-WTP disparity (see also Kling et al., 2013). Also this uncertainty effect is not uncontested (see Davis and Reilly, 2012).

A second explanation from prospect theory is that people value losses more than gains, which has come to be known as loss aversion (Kahneman and Tversky, 1979). This hypothesis has been tested in various fields, such as marketing (e.g., Wertenbroch et al., 2007), transport economics (e.g., Li and Hensher, 2011), health economics (e.g., Viscusi and Huber, 2012; Chilton et al., 2012) and environmental economics (Mansfield, 1999; Koetse and Brouwer, 2013).

The third group of explanations for the observed disparities are provided by the literature on the effects of bounded rationality, decision heuristics and decision anomalies, on choices made in hypothetical markets (for an overview see Leong and Hensher, 2012). For example, a study by Bateman et al. (2009) shows that using virtual reality instead of the standard representations of choice attributes in choice experiments may reduce the WTA-WTP gap substantially. They show that advanced disclosure of choice attributes and levels reduces respondent judgment error and moderates reliance on loss aversion in uncertain situations. There are several other studies that provide evidence that experience and training mitigate or even fully eradicate the observed WTA-WTP disparity, thereby questioning the relevance of loss aversion and prospect theory (e.g., Kling et al., 2013; Bateman et al., 2008; Plott and Zeiler, 2005; List, 2003).

In this paper we focus on an issue that belongs to the third category, i.e., attribute non-attendance in choice experiments. Specifically, we analyse whether differences in attribute attendance between WTP and WTA experiments can contribute to an explanation of the WTA-WTP disparity. In non-market valuation of public goods the payment vehicle used is often some form of tax; WTP estimates are obtained by presenting people with tax increases, WTA estimates are obtained by presenting people with tax decreases. However, where the occurrence of tax increases in reality are likely perceived as credible, the occurrence of tax decreases in reality are likely not, or far less so. In choice experiments people may therefore ignore the tax attribute more in case of tax decreases than in case of tax increases. When an individual does not attend to the tax attribute this basically means a zero tax coefficient for that individual, which artificially decreases the overall tax coefficient estimate and artificially increases welfare estimates. Higher non-attendance for tax decreases than for tax increases therefore gives rise to and/or increases the WTA-WTP disparity.

Our main hypotheses are therefore that controlling for tax non-attendance (1) substantially reduces value estimates from choice experiments, and (2) substantially reduces the WTA/WTP disparity. We test these hypotheses by comparing results of standard choice models with those of attribute non-attendance models, using data from two choice experiments in the Netherlands that contain both WTP and WTA choice questions.

4.2.2 Attribute non-attendance in choice experiments

The *rational* behavioural model assumes that individuals process and use all information available to them to determine their preferences and make their choices. In contrast, the *rationally-adaptive* behavioural model assumes that individuals know that their information processing abilities are limited, and allocate their attention to the information provided such that costs of cognition are minimized and benefits of information processing are maximized (DeShazo and Fermo, 2004). The latter model induced the development of an approach that has become known as attribute non-attendance. Initial applications of this approach are provided in Hensher (2007), Campbell et al. (2011) and Hensher et al. (2012). The attribute non-attendance approach assumes that individuals pay an unequal amount of attention to the choice attributes used in a choice experiment. They pay more attention to attributes that are considered to be more important, and much less attention to, or even ignore, attributes that are considered to be less important. This may be due to a various factors, such as time pressure, cognitive load, task complexity and attribute or attribute level credibility (e.g., Saelensminde, 2006; Hensher, 2007; Hensher et al., 2012). The most problematic consequence of attribute non-attendance is that, without accounting for a limited trade-off between attributes, the marginal rates of substitution and welfare estimates are biased (Puckett and Hensher, 2008; Campbell et al., 2008).

Empirical studies show that accounting for attribute non-attendance in choice models results in a much better fit, but evidence on its effects on welfare estimates is somewhat mixed. While most studies find a substantial decrease in welfare estimates (Campbell et al., 2008; Scarpa et al., 2012; Hensher and Greene, 2010; Scarpa et al., 2009; Puckett and Hensher, 2008), some studies find an increase (e.g., Hensher et al., 2007). The underlying processes here can be twofold. As is well-known, the WTP associated with an attribute or attribute level *A* is given by:

$$\mathrm{WTP}_A = -\frac{\beta_A}{\beta_M}, \tag{1}$$

where β_A is the coefficient on attribute *A* and β_M is the coefficient on the monetary attribute. for those respondents that do not attend to the monetary attribute it holds that $\beta_M = 0$. Not accounting for this in, for example, a MNL model invariably pushes β_M downward in absolute terms, and thereby *increases* WTP_A. In the same fashion, non-attendance to attribute *A* invariably pushes β_A downward in absolute terms, and thereby *decreases* WTP_A. When both non-attendance to the monetary attribute and attribute *A* occur, the net effect on WTP_A is ambiguous and thereby case specific. In this paper we look specifically at non-attendance to the monetary attribute, both in WTA and in WTP questions. In our study WTA and WTP are elicited within the same experiment, implying attribute non-attendance to other attributes is likely the same for WTA and WTP. Effects of non-attendance to other attributes therefore likely cancel out when looking at the WTA-WTP ratio.

Two approaches for detecting and correcting for attribute non-attendance exist. The first approach makes use of stated non-attendance, for which individuals need to indicate explicitly whether or not they ignored certain attributes in the choice tasks presented to them. The second approach infers non-attendance from the data, using models to obtain information on the existence and extent of attribute non-attendance. Ultimately, the attribute parameters are weighted according to individual stated or inferred attribute non-attendance information. In this paper we make use of inferred non-attendance, because it produces more robust results and because stated non-attendance faces several problems that are related to erroneous reporting by respondents (Hess and Hensher, 2010; Carlsson et al., 2010; Alemu et al., 2013; Scarpa et al., 2012).

There are two methods for inferring non-attendance from choice data. The first approach makes use of the coefficient of variation of individual-specific posterior means and variances, which are based on random parameter mixed panel logit models (e.g., Hess and Hensher, 2010; Scarpa et al., 2012). The second approach, which we use in this paper, is estimating an equality-constrained latent class (ECLC) model, developed by, among others, Campbell et al. (2010) and Scarpa et al. (2012). The ECLC approach is a two-step procedure. The first step is estimating a latent class model, with the restrictions that ignored attributes have coefficients equal to zero and that coefficients for attended attributes are the same across all classes. These restrictions ensure that the estimated class probabilities actually reflect attribute non-attendance rather than preference heterogeneity between classes. At the second step, either the estimated coefficients are used to calculate welfare values (Hensher et al., 2012), or the estimated class probabilities are used to weight the attribute coefficients in the MNL model so as to account for individual degrees of attribute non-attendance (Campbell et al., 2010; Scarpa et al., 2012).

4.2.3 Design of the experiments

Experiment 1

The purpose of the first choice experiment is to assess consumer preferences for different types of natural areas and their characteristics, and for this we present respondents with choice alternatives that are of a generic nature instead of referring to a specific site or area. From the literature it is evident that consumer preferences for natural areas may be affected by many characteristics. In this experiment we include the type of natural area, its size, the distance from the residence to the area, accessibility of the area, degree of fragmentation, and changes in annual municipal tax, which is an annual tax in the Netherlands that is levied separately from national income taxes. For details on this experiment we refer to Koetse et al. (2014). In order to be able to estimate both WTP and WTA estimates, we include tax increases and tax decreases, and use five levels for each. A summary of attributes and attribute levels is provided in Table 5.

Table 5. Attributes and attribute levels in experiment 1

Attributes	Number of levels	Levels
Type of natural area	3	▪ Small scale grassland ▪ Natural area with water ▪ Forest
Size of natural area	3	▪ 2 km^2 ▪ 6 km^2 ▪ 16 km^2
Distance to natural area	3	▪ 1 km ▪ 5 km ▪ 15 km
Fragmentation of natural area by urban sprawl	3	▪ No fragmentation (1 patch) ▪ Medium fragmentation (2 patches) ▪ High fragmentation (4 patches)
Fragmentation of natural area by transport infrastructure	3	▪ No fragmentation ▪ Medium fragmentation (one road) ▪ High fragmentation (two roads)
Accessibility for recreation	2	▪ Accessible ▪ Not accessible
Annual municipal tax	5	▪ Increase/decrease of € 320,- ▪ Increase/decrease of € 140,- ▪ Increase/decrease of € 60,- ▪ Increase/decrease of € 20,- ▪ No change

In presenting the choice options we use text to describe the attribute levels, and below the text we include figures for graphical representation. These figures represent all attributes except for the payment vehicle. An example choice card is shown in Figure 12. In the choice cards we do not include an opt out alternative (i.e., giving the respondent the option to state they do not know which alternative they prefer) or a status quo alternative because we aim to identify generic relative preferences for natural areas and their characteristics in the Netherlands, rather than to identify whether consumers prefer change to no change in a specific situation.

The attributes and their levels can be combined to generate (3×3×3×3×2×5=) 810 possible choice alternatives. These obviously cannot all be shown to respondents, hence a fractional factorial statistical design was generated using the Sawtooth CBC software. The program uses a randomised design strategy, and produces a design that is nearly as orthogonal as possible within respondents (i.e., correlation between attribute levels for each respondent is minimal). We generate a statistical design containing 100 survey versions of 12 choice tasks each.

For data collection we used a Dutch internet panel managed by TNS-NIPO, containing over 200,000 households. The panel is established through random sampling, meaning that each member of society has an equal chance to be added to the panel as long as he or she has conveyed the willingness to cooperate. Throughout the entire data collection process, respondents were sampled using representative sampling (for the entire Dutch population) on age, gender, education, household size and size of municipality. A total of 2,100 questionnaires were sent out, and ultimately 1,360 complete responses were obtained, implying a response rate of nearly 65%. For the choice experiment we have a total of 16,102 observations.

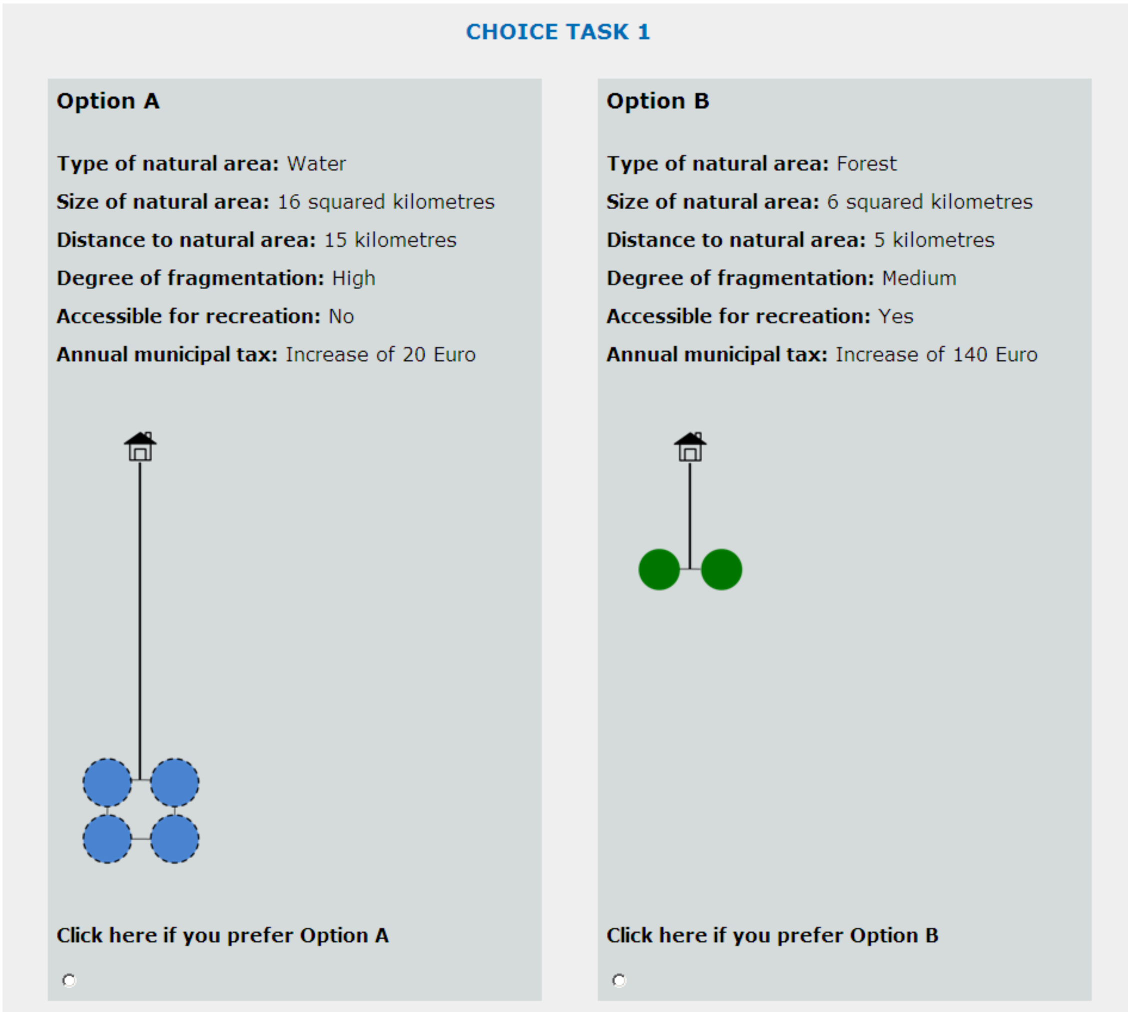

Figure 12. Choice card example experiment 1 (all lines and circles are drawn to scale)

Experiment 2

The second experiment is the study that has been discussed in Section 4.1 of this deliverable. For this paper we use two of the four subsamples. The attributes and attribute levels for the WTP and WTA experiment are summarised in Table 6.

For each sample we employed representative sampling based on age, gender, household size, social class (education and profession) and residential location. For each choice experiment, 375 households were invited to complete the survey. The total number of complete respondents for the

WTP experiment is 291, while for WTA it is 299, yielding a response rate of around 80 percent. The socio-demographic characteristics of the two samples reveal that the samples are very similar in composition. For all other issues we refer to Section 4.1.

Table 6. Attributes and attribute levels experiment 2

Attributes	Attribute levels
Raising the dikes	▪ No (reference scenario in WTP version) ▪ Yes (reference scenario in WTA version)
Flood probability	▪ Once every 10,000 years (reference scenario WTA2) ▪ Once every 2,000 years ▪ Once every 1,000 years (reference scenario WTP2)
Change in bird population	▪ 0 % (reference scenario WTA) ▪ –10 % ▪ –30 % (reference scenario WTP)
Type of shore	▪ No shores (reference scenario WTP) ▪ Shores off the coast ▪ Shores next to the dike (reference scenario WTA)
Change in annual local tax (increases for WTP, decreases for WTA)	▪ €0 (reference scenario) ▪ €60 ▪ €100 ▪ €180

4.2.4 Estimation results and value estimates

As discussed in Section 2 we make use of inferred attribute non-attendance and estimate an equality-constrained latent class (ECLC) model. In the model, size of the area, distance to the area and tax are included in the model as continuous variables, while attribute levels for type, fragmentation and accessibility of the natural area are dummy coded (grassland, no fragmentation and accessible areas are used as reference categories). We estimate and compare results from MNL and ECLC models. We estimate models on a dataset containing both WTA and WTP questions, and estimate separate coefficients for tax increases and decreases. For the ECLC model we focus only on tax non-attendance. Although non-attendance to other attributes is potentially relevant as well, it is likely equally relevant for both WTA and WTP questions. The effects of non-attendance to other attributes therefore cancel out when looking at the WTA-WTP ratio, which is our measure of interest.

Estimation results experiment 1

For experiment 1 we estimate an ECLC model with four classes; class 1 without constraints on the tax coefficients, class 2 with the tax decrease coefficient constrained to zero, class 3 with the tax increase coefficient constrained to zero, and class 4 with both tax coefficients constrained to zero. Apart from these tax coefficient restrictions, all coefficients are constrained to be identical for all classes (including non-zero tax coefficients). MNL and ECLC estimation results for experiment 1 are presented in Table 7. With the exception of the tax coefficients, the signs, magnitudes and statistical significance of all other attribute coefficients are very similar for the two models. In both models water and forest are preferred to grasslands, size of an area increases and distance to an area decreases its value, medium fragmentation has a small negative effect while strong fragmentation has a substantial negative effect on the value of an area, and also when an area is not accessible for recreation its value decreases substantially (for details see Koetse et al., 2014). The first difference between the two models is that the ECLC model has a higher explanatory power than the MNL model. The second difference is that the estimated tax coefficients are substantially higher in absolute terms in the ECLC than in the MNL model. Another interesting finding is with respect to the estimated latent class probabilities. These results show that only 12% of respondents attend to both tax increases and decreases, that around 55% of respondents do not attend to tax decreases but do attend to tax increases, that non-attendance to tax increases only is almost non-existent, and that around 32% of respondents do not attend to both tax increases and decreases. Clearly, non-attendance to the tax attributes is large, and is substantially larger for tax decreases than for tax increases.

Table 7. MNL and ECLC estimation results for experiment 1 (ECLC tax coefficients that are constrained to zero are not shown)

Attribute variables	MNL beta	MNL s.e.	ECLC beta	ECLC s.e.
Water (dummy; ref = grass)	0.237**	0.027	0.261**	0.029
Forest (dummy; ref = grass)	0.657**	0.028	0.772**	0.031
Size (continuous in km^2)	0.020**	0.002	0.023**	0.002
Distance (continuous in km)	–0.031**	0.002	–0.034**	0.002
Fragmentation: Medium (dummy; ref = low)	–0.056*	0.027	–0.068*	0.030
Fragmentation: Strong (dummy; ref = low)	–0.400**	0.028	–0.424**	0.031
Area not accessible (dummy; ref = accessible)	–0.698**	0.026	–0.769**	0.029
Tax increase (continuous in 100 Euro)	–0.917**	0.024	–2.223**	0.124
Tax decrease (continuous in 100 Euro)	0.151**	0.015	4.048**	0.688
Prob. class 1 (No ANA)	--	--	0.120**	0.014
Prob. class 2 (ANA to tax decreases only)	--	--	0.556**	0.021
Prob. class 3 (ANA tax increases only)	--	--	0.007	0.004
Prob. class 4 (ANA to both tax in- and decreases)	--	--	0.317**	0.021
NOBS	16,102		16,102	
Log–L	–9,141		–8,869	
Pseudo R2 (adjusted)	0.181		0.205	

**, * = statistically significant at 1% and 5%, respectively

Estimation results experiment 2

For experiment 1 we estimate an ECLC model with three classes; class 1 with the tax decrease coefficient constrained to zero, class 2 with the tax increase coefficient constrained to zero, and class 3 with both tax coefficients constrained to zero. Apart from these tax coefficient restrictions, all coefficients are constrained to be identical for all classes (including non-zero tax coefficients).

The reason for not including a class without restrictions on both tax coefficients is that this class does not exist. The reason is that WTP and WTA estimates were obtained from separate samples. Including this class does not change estimation results, but is does change class probabilities and renders estimating sensible class probability standard errors impossible. MNL and ECLC estimation results for experiment 2 are presented in Table 8. All coefficients for both models have the expected signs and are statistically significant at least at 5%. Increasing flood probabilities decrease utility, crating shores along the coast have a positive effect on preferences, and decreases in bird population are valued negatively. Very similar to experiment 1, the ECLC model has a higher explanatory power than the MNL model. Also the estimated tax coefficients are substantially higher in absolute terms in the ECLC than in the MNL model, and more so for tax decreases than for tax increases. The interpretation of the latent class probabilities is a little trickier for this experiment, since we not have WTA and WTP observations for each individual. This means that, by definition, for each individual one of the tax coefficients is equal to zero (which is also why we did not include a fourth class containing two unrestricted tax coefficients). Since the sample sizes of the WTP and WTA experiment are almost equal, one would expect the class probabilities for class 1 and 2 to be comparable in size when there is no difference in non-attendance to tax decreases and increases. Clearly, however, these class probabilities are not equal, and the percentage of respondents that do not attend to tax decreases is more than two times larger than for tax increases. In conclusion, as for experiment 1, non-attendance to the tax attributes is large, and is substantially larger for tax decreases than for tax increases.

Table 8. MNL and ECLC estimation results for experiment 2 (ECLC tax coefficients that are constrained to zero are not shown)

	MNL		ECLC	
Attributes	**beta**	**s.e.**	**beta**	**s.e.**
No increase in dike height (label)	0.337**	0.063	0.426**	0.073
Flood probability 1 in 1,000 → 1 in 2,000 (d)	0.426**	0.054	0.583**	0.058
Flood probability 1 in 1,000 → 1 in 10,000 (d)	0.886**	0.051	1.151**	0.062
No shore → Off the coast (d)	0.307**	0.052	0.430**	0.058
No shore → Next to dike (d)	0.425**	0.052	0.605**	0.062
Bird population –30% → –10% (d)	0.675**	0.054	0.861**	0.058
Bird population –30% → 0% (d)	1.018**	0.054	1.281**	0.059
Tax increase (continuous in Euro)	-0.0121**	0.0006	-0.0353**	0.0012
Tax decrease (continuous in Euro)	0.0015*	0.0006	0.0223**	0.0009
Prob. class 1 (ANA to tax decreases only)	--	--	0.474**	0.033
Prob. class 2 (ANA tax increases only)	--	--	0.224**	0.057
Prob. class 3 (ANA to both tax in- and decreases)	--	--	0.302**	0.058
NOBS	5,960		5,960	
Log–L	–5,512		–4,563	
Pseudo R2 (adjusted)	0.096		0.302	

**, * = statistically significant at 1% and 5%, respectively

Value estimates and the WTA-WTP disparity

Value estimates for experiments 1 and 2 are shown in Table 9, which presents WTP and WTA estimates for the MNL and ECLC models. When comparing average ECLC results with MNL results for experiment 1, WTP estimates decrease by around 30% while WTA estimates decrease by around 50%. When comparing average ECLC results with MNL results for experiment 2, WTP estimates decrease by around 55% while WTA estimates decrease by around 90%. In conclusion, not accounting for non-attendance to the tax attribute in choice experiments may clearly lead to substantial overestimation of value estimates.

Table 9. WTA and WTP estimates from MNL and ECLC models for experiments 1 and 2

Attributes	WTP MNL	ECLC		WTA MNL	ECLC	
Experiment 1						
Water (dummy)	€ 26	€ 17	–33%	€ 157	€ 51	–55%
Forest (dummy)	€ 72	€ 51	–28%	€ 435	€ 150	–52%
Size (in km^2)	€ 2	€ 2	–30%	€ 13	€ 4	–53%
Distance (in km)	–€ 3	–€ 2	–33%	–€ 21	–€ 7	–55%
Fragmentation medium (d)	–€ 6	–€ 5	–26%	–€ 37	–€ 13	–50%
Fragmentation strong (d)	–€ 44	–€ 28	–35%	–€ 265	–€ 82	–56%
Inaccessible (d)	–€ 76	–€ 51	–33%	–€ 462	–€ 150	–55%
Experiment 2						
No increase in dike height (label)	€ 28	€ 12	-57%	€ 232	€ 19	-92%
Flood probability 1 in 1,000 → 1 in 2,000 (d)	€ 35	€ 17	-53%	€ 293	€ 26	-91%
Flood probability 1 in 1,000 → 1 in 10,000 (d)	€ 73	€ 33	-56%	€ 609	€ 52	-92%
No shore → Off the coast (d)	€ 25	€ 12	-52%	€ 211	€ 19	-91%
No shore → Next to dike (d)	€ 35	€ 17	-51%	€ 292	€ 27	-91%
Bird population –30% → –10% (d)	€ 56	€ 24	-56%	€ 464	€ 39	-92%
Bird population –30% → 0% (d)	€ 84	€ 36	-57%	€ 699	€ 57	-92%

Non-attendance is substantially larger for tax decreases than for tax increases, and because of this the changes in value estimates are larger for WTA than WTP. As a result the WTA-WTP disparity decreases substantially when non-attendance to tax is taken into account (see Figure 13). Where WTA estimates are around 6 and 8 times higher than WTP estimates for the MNL models, this ratio decreases to around 3 for the non-attendance models.

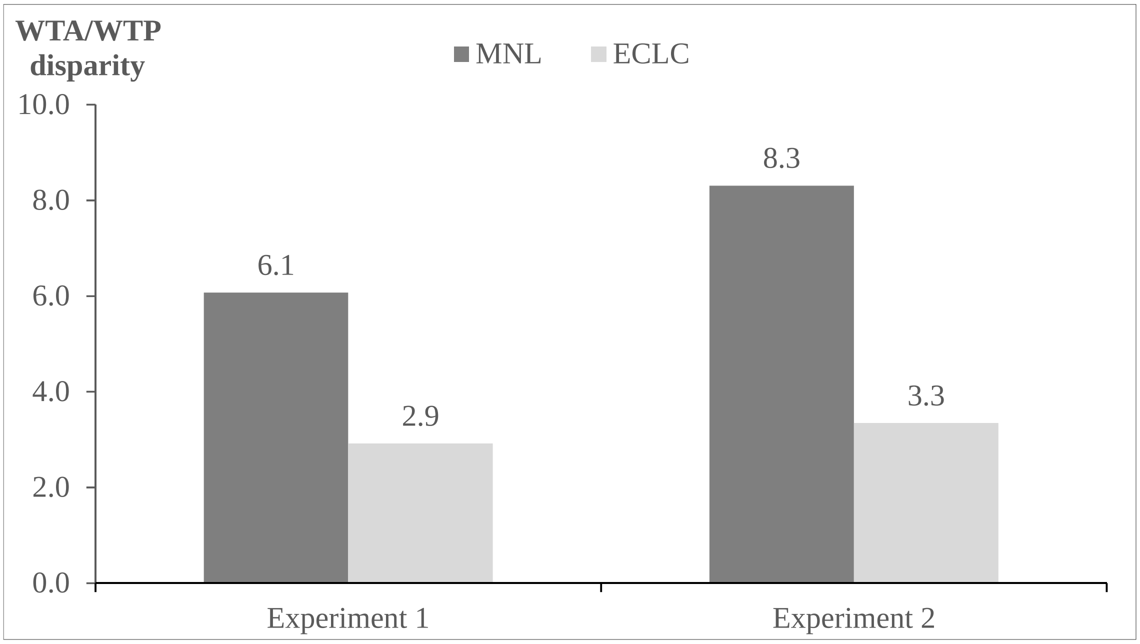

Figure 13. WTA-WTP ratios (on the y-axis) for the MNL and the ECLC model

4.2.5 Conclusions and discussion

Using two choice experiments on the valuation of ecosystem services that contain both WTP and WTA questions, we show that non-attendance to the tax attribute in choice experiments is substantial, and pushes down the tax coefficient. Since tax is a logical and often used payment vehicle in choice experiments aimed at ecosystem service valuation, this result suggests that ecosystem service value estimates obtained from choice experiments will generally overstate true values by a substantial amount when non-attendance is not taken into account (see also previous section). We also show that non-attendance to tax decreases is larger than for tax increases. This means that, when tax non-attendance is controlled for, decreases in WTA are larger than decreases in WTP, and that the WTA-WTP disparity decreases substantially. Specifically, the ratio goes down from 6 to 8 in standard MNL models, to around 3 in non-attendance models. These values are far more in line with disparities found for revealed preference experiments (see Horowitz and McConnell, 2002).

Our findings have several implications for using choice experiments for the valuation of ecosystem services, and public goods in general. First, both WTP and WTA decrease substantially when controlling for non-attendance to the monetary attribute, especially when tax is used as a choice attribute. Non-attendance models could therefore be used alongside other, more traditional models in order to obtain more conservative, and in our opinion more reliable welfare estimates. Second, non-attendance to tax decreases appears to be much larger than for tax increases. Under normal circumstances this leads to substantially larger upward bias in WTA than in WTP, thereby strongly increasing the WTA-WTP disparity. Since tax is often the most logical monetary attribute in choice experiments aimed at ecosystem service valuation, and since the WTA is in many situations a more logical measure of welfare than the WTP, our findings are both worrisome and hopeful. Worrisome because a large part of respondents ignores tax decreases, implying that many observations are needed in order to obtain representative and reliable WTA estimates. Hopeful because the recent development of models to control for non-attendance makes it possible to filter out these ‘anomalies’ in stated choice behaviour, and makes it possible to get more accurate WTA estimates, which can be used in actual welfare calculations.

An important discussion with respect to the interpretation of attribute non-attendance is on whether it actually reflects non-importance (e.g., Hess et al., 2013). Stated differently, non-attendance to the monetary attribute could in fact reflect that for some respondents the changes in tax are too small to be of interest. Although this issue is not explicitly addressed in this paper, we consider it unlikely that this is the case in our study, let alone that it can explain the patterns in our findings. There are two reasons for this. First, the changes in tax presented to respondents are relatively large, making it unlikely that people systematically ignore this attribute because the changes would have no effect on their utility. Second, and more importantly, in experiment 1 each respondent was shown both WTP and WTA questions, and ignored tax decreases far more often than tax increases. In fact, there were (almost) no respondents that ignored tax increases while not ignoring tax decreases. This shows that respondents in this experiment did consider the changes in tax important for their utility, and ignored tax decreases more often than tax increases because of other reasons.

Finally, it is interesting to assess whether other case studies produce similar findings, but even more important is to analyse whether using other payment vehicles produce the same patterns. Relevant associated questions are whether other payment vehicles are ignored less often, whether the gap between decreases and increases is smaller, and whether they produce more reliable estimates of welfare change. If this is indeed the case, it would make these payment vehicles experimentally more suitable, if not always more logical in reality, for measuring monetary value changes through choice experiments.

Another issue is that of measuring the effects of stated non-attendance, i.e., asking respondents whether they attended to an attribute or not, rather than of inferred non-attendance through choice models. The two experiments discussed in this section did not ask this question specifically enough to address this issue. In the next section we discuss the outcomes of a study that did ask this question in a very specific way, and, among other things, we address the effects of stated non-attendance on the results. A related issue is that of comparing the outcomes of models that correct for inferred non-attendance and of models that correct for stated non-attendance. More specifically, the issue is whether stated and inferred non-attendance have similar effects on estimation results, or do they produce different patterns. We leave this issue for further research.

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4.3 The effects of the price attribute on trade-offs and choice in a choice experiment[11]

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4.3.1 Introduction

The importance of cultural ecosystem services in agrarian landscapes is increasingly recognized as the quality of many landscapes is affected by scale enlargement and intensification of agricultural practices. Parallel to these processes of landscape change, there is a growing societal demand for cultural services in many landscapes (Sayer et al., 2013; Zasada, 2011). In many landscapes, scale enlargement and intensification lead to negative welfare effects. The notion that cultural and recreational landscape qualities should be protected through European agricultural policies by compensating farmers for landscape conservation and maintenance is gaining traction in science and policy (Plieninger et al., 2013; van Zanten et al., 2014a).[12] However, developing robust methods to identify the relative values of agrarian landscape attributes in a consistent way remains a challenge (Schaich et al., 2010).

To grasp the value of landscapes and to identify which landscape attributes contribute to this value, many studies have investigated stated preferences for agricultural landscapes. Here, we can distinguish between holistic landscape assessments and attribute-based studies that focus on specific characteristics of landscapes (Hynes et al., 2011). Both holistic and attribute-based studies have applied a diverse portfolio of elicitation methods, including several economic valuation methods (Campbell, 2007; Grammatikopoulou et al., 2012) and non-economic approaches (Arriaza et al., 2004; Dramstad et al., 2006). Economic valuation methods, such as contingent valuation and choice modeling, estimate the willingness to pay (WTP) of respondents for landscape scenarios or attributes by including a financial trade-off. Non-economic landscape preferences – in the context of ecosystem services often referred to as socio-cultural values (de Groot et al., 2010; Millennium EA, 2005) – are measured using a heterogeneous portfolio of conceptual approaches and assessment methods (Van Zanten et al., 2014b).

[11] This section is based on Van Zanten BT, Koetse MJ, Verburg PH, 2015, Economic valuation at all cost? The role of the price attribute in a landscape preference study, Discussion Paper, Institute for Environmental Studies, VU University Amsterdam, the Netherlands (under review at *Ecosystem Services*).
[12] The Treaty on the Functioning of the European Union promotes the polluter pays principle, alongside principles of precaution, prevention and rectifying pollution at source in European law (European Union, 2012). This is not at odds with the principle of compensation, in fact, the two principles can be compensatory, e.g., when compensations are financed by polluters.

Both economic and non-economic approaches have their pros and cons. Economic valuation of ecosystem services is often contested in the literature (e.g., Chan et al., 2012). Discussions on the use of monetary valuation of ecosystem services range from objections against commodification of the environment (Gomez-Baggethun and Ruiz-Perez, 2011; Schröter et al., 2014) to the risk of inaccurate value estimates as a result of the limited understanding of the functioning of highly complex ecological systems (Kumar and Kumar, 2008). Note, however, that the first criticism is partly aimed at the approach itself, but also at the use of the results, while the second criticism also includes non-economic approaches to values and preferences (although deliberative approaches do allow people to learn and to explore the uncertainties involved, while valuation survey generally do not; see also Chapter 3 of this deliverable). In addition, measuring preferences and values through choice experiments has its specific methodological constraints, since preference estimates have been found sensitive to both the predefined levels of the attributes in the experiment and attendance level of the payment vehicle by respondents (Hess et al., 2013; Scarpa et al., 2009). Of course, the first criticism also holds for non-monetary choice experiments. Socio-cultural valuation methods that are used to measure landscape preferences and ecosystem services also have their methodological constraints (Daniel et al., 2012; Schaich et al., 2010). The absence of a trade-off between landscape attributes in the design of many landscape preference studies makes it hard to distinguish the relative preferences different attributes. Moreover, not including a payment vehicle in experiments makes it impossible to express preferences in monetary terms, implying that an overview of the full costs and benefits is difficult to obtain, allowing for implicit rather than explicit trade-offs in decision making.

Given the potential limitations of obtaining accurate *monetary value estimates* through choice modeling, but acknowledging the advantages of choice modelling over other methods that do not allow for trading off different landscape attributes, using choice experiments without a monetary payment vehicle may be an interesting alternative to measure preferences. A number of studies in landscape research applied such an approach to explore preference heterogeneity among groups of beneficiaries (Arnberger and Eder, 2011; Soini et al., 2012). Outside landscape research, Aas et al. (2000) and Wattage et al. (2005) applied non-monetary choice modeling to assess the relative importance of attributes of fisheries management schemes. However, these studies did not test whether relative preferences for landscape attributes will change as a result of adding a payment vehicle. The relevant associated question is whether people respond differently to choice situations when a financial trade-off is included. The literature on this issue is limited, but the available evidence shows that there are significant differences in the marginal rate of substitution (MRS) and the ranking of individual attributes between experiments with and experiments without a monetary payment vehicle (Aravena et al., 2014; Carlsson et al., 2007). These results contradict the assumption made in mainstream economics that relative preferences for attributes are additively separable and are independent of the inclusion of a payment vehicle. Carlsson et al. (2007) suggest that the differences are caused by cognitive overload as a result of increased complexity of the experiment due to adding a payment vehicle. In addition to these factors, there is the possibility that people do not include the payment vehicle in making their choices (payment vehicle non-attendance), either due to cognitive overload or due to other factors such as protest votes (rejecting the notion of paying through a price or a tax). Scarpa et al. (2009), for example, found in an image-based rural landscape preference study that 80-90 percent of the respondents ignored the payment vehicle. This could affect study outcomes with respect to relative preferences for landscape attributes, i.e., people who ignore the price may make different choices.

The objective of this study is to address the effect of including a payment vehicle on relative preferences for landscape attributes. We apply an image-based choice experiment to address preferences for attributes of agricultural landscapes in a case study area in the Eastern part of the Netherlands. We apply a split-sample approach in which about half of the respondents completed a choice experiment without a payment vehicle, and the other half completed a choice experiment with a payment vehicle. First, we compare the outcomes of Multinomial logit (MNL) model estimates for the two samples. Second, we assess the effect of non-attendance of the payment vehicle on relative preferences for landscape attributes and WTP estimates in the monetary experiment. Here, we test the intuitive hypothesis that respondents who ignore the payment vehicle and respondents in the non-monetary experiment express similar landscape preferences.

4.3.2 Data collection and research design

In July 2013, a total of 425 questionnaires were completed through face-to-face interviews in the Dutch municipality of Winterswijk. The respondents, who were all tourists, were interviewed on tourist accommodations such as campsites, bed & breakfasts and hotels. Tourist accommodations were selected randomly. A total of 191 respondents completed the non-monetary experiment vehicle, while 234 respondents participated in the monetary experiment with payment vehicle. Within the sample of the monetary experiment, we make a distinction between a stated non-attendance sample and a stated attendance sample. The stated non-attendance sample contains respondents who indicated that they did not take into account the price in the choice process, either because of cognitive overload or because of other reasons such as rejecting the notion of paying for a more attractive landscape through an increase in overnight prices. The payment vehicle is defined as the extra costs respondents would have to pay per overnight stay per room/tent. Although this means that total costs are higher for those people that stay longer, the percentage increase in costs is still identical for all respondents. We intentionally avoided the term tourist tax for the payment vehicle, since there had been negative publicity about the local spending of tourist taxes and many respondents anchored preferences on tourist tax during the pre-test phase. Except for the inclusion of a payment vehicle in the monetary choice experiment, questions in all 425 surveys were identical.

Regarding the key demographic variables age, education level, gender and income, the samples of the non-monetary and the monetary experiment are almost identical. Table 10 shows that in both samples, the mean year of birth is 1954. The mean and mode education level class in both samples is 'vocational'. Also, both samples contain slightly more female respondents (53% and 52% female). The mean income class in both samples is a net household income of 2000-2500 euros per month. There is no secondary statistical data available to validate the representativeness of our samples in terms of these key demographic variables. However, the fact these samples are rather identical indicates that we have drawn a representative sample of the visitor population that was present during the period of our data collection and on the selected interview locations.

Table 10. Demographic background variables of the sampled population in both experiments

	Non-monetary experiment (n=191)			**Monetary experiment (n=234)**		
Demographic characteristics	*Non-response*	*Mode*	*Mean*	*Non-response*	*Mode*	*Mean*
Year of birth	6.81%	1945	1954	2.99%	1952	1954
Education class	0.00%	Vocational	Vocational	0.85%	Vocational	Vocational
Gender	0.00%	Female(1)	0.53	0.00%	Female(1)	0.52
Income class	12.04%	2000-2500	2000-2500	14.10%	1500-2000	2000-2500

The survey is designed to provide policy relevant information of tourist preferences for the visual quality of the agricultural landscape. The type of landscape attributes that are included in the experiment were selected based on a meta-analysis of European landscape preference studies (van Zanten et al., 2014b) and the visual appearance of these attributes in the local landscape context was further specified in close collaboration with a focus group of local landscape experts. The landscape alternatives were presented to the respondents using digitally calibrated images instead of the standard tabular format of choice cards. Results by Bateman et al. (2009) show that an image-based approach enhances the evaluability of choice situations, thereby reducing respondent judgment error. Additionally, the use of photographs as a valid surrogate for a real landscape experience has been accepted and is common practice in non-economic landscape preference studies since several decades (Ode et al., 2009).

First, we tested the digitally calibrated landscape images for clarity with academic colleagues. Second, the choice experiments with and without payment vehicle were each pretested on 50 respondents in the case study area. Subsequently, we estimated MNL models for both experiments based on the collected data in the pretest, the results of which served as priors for generating two separate efficient statistical designs.

The final questionnaire consists of three sections. The first section includes motivations for the visit to the area and characteristics of visit (e.g., length of stay, activities). The second section contains the choice experiment and questions addressing attribute (non-)attendance. The third part contains socio-demographic background characteristics of respondents. Each questionnaire includes eight choice cards. For both the monetary and the non-monetary experiment, the design consists of six blocks (versions).

Table 11 presents the attributes and their levels. With the landscape alternatives, composed of combinations of the four landscape attributes, we aim to capture the relevant variations in landscape preferences related to landscape structure and composition in the case study area and thus provide useful information for the local landscape management and targeting of agri-environmental policy measures. Therefore, the attribute levels of the maize-grassland ratio (representing the variation of agricultural land uses in the area), prevalence of hedgerows and tree lines and prevalence of forest patches are based on their actual occurrence in the study region. For those attributes, the mid-level represents the landscape average while the high-level is chosen in the upper part of the frequency distribution of occurrence in the study region. The presence of livestock is defined as a dichotomous variable. For all attributes, the low level is defined as the absence of the landscape attribute in the visualization.

Table 11. Attributes and levels

Attribute	Level
Presence of livestock	0) no presence of livestock 1) presence of livestock
Maize-grassland ratio	0) low 1) landscape average 2) high
Prevalence of hedgerows and tree lines	0) low 1) landscape average 2) high
Prevalence of forest patches	0) low 1) landscape average 2) high
Extra costs per overnight stay per room/tent	1 euro; 2,50 euro; 5 euro; 10 euro

Figure 14 shows an example of a choice card with visualized landscape alternatives. All landscape alternatives are visualized in a base landscape image that is representative for the case study area in terms of land cover structure and composition in the background and biophysical characteristics (e.g., flat terrain) as proposed by Appleton and Lovett (2003) and Arnberger and Eder (2011). Within this base landscape image, the four landscape attributes are visualized. Alternative A shows a landscape without livestock, a landscape average maize-grassland ratio, a low level of hedgerows and tree lines and a high level of forest patches. Similarly, alternative B shows no livestock and an average maize-grassland ratio, but in contrast to alternative A, a high level of hedgerows and tree lines and a low level of forest patches. In landscape alternative C, there is a medium level of hedges, maize-grassland ratio and forest patches and there is livestock present. The images were digitally calibrated using Adobe Photoshop and all attributes were visualized in a way that they could be varied independently.

Figure 14. Example of a choice card; all alternatives were presented in A4 paper format to the respondents

4.3.3 Model specification and comparative analysis

For the analysis of the responses, we estimate a MNL choice model for both experiments. Although many complex choice models exist, the multinomial logit (MNL) is still the starting point for any choice modelling analysis (Louviere et al., 2003). In our choice model, each attribute level is dummy-coded, except for the price attribute in the monetary experiment, and the minimum levels of all attributes are included as the reference category.

We have analyzed the influence of including a payment vehicle by comparing the MRS of the attribute levels, following the approach of Aravena et al. (2014) and Carlsson et al. (2007). Comparing the MRS has enabled us to assess the relative importance of the landscape attributes in both the monetary and the non-monetary experiment. In addition, for the monetary experiment we have also considered the relation between stated (non-)attendance of the payment vehicle and relative preferences for the landscape attributes. Subsequently, we have compared the MRS of the landscape attribute levels of the non-monetary experiment to the MRSs of sub-samples of respondents of the monetary experiment who indicated that they did or did not include the price in their choices. To illustrate the significance of price attribute non-attendance, we also show the effects of stated price attribute non-attendance on WTP estimates of the (non-)attendance sub-samples and the full sample of the monetary experiment.

Comparing the relative importance of landscape attributes

Table 12 displays the output of the MNL models for the samples with and without payment vehicle. The MNL model for the sample without payment vehicle was estimated based on 191 completed questionnaires and has an adjusted R^2 of 0.28. All estimated landscape attribute coefficients are statistically significant ($p < 0.05$). The MNL model for the sample with payment vehicle has an adjusted R^2 of 0.23 and a slightly larger sample size, consisting of 234 respondents. Here, all attribute coefficients except for a medium prevalence of forest patches, are statistically significant ($p < 0.05$). As a result of possible differences in the scale parameter, coefficients of the two models cannot be compared directly. Because the two experiments do not have the same structure (i.e., one of the experiments includes a payment vehicle), we cannot perform the Swait and Louviere test (Swait and Louviere, 1993) to assess whether heterogeneity is due to differences in preferences or due to a difference in the scale parameter.

Comparing the ranks of the individual attributes and their levels provides a first, rather coarse, indication of the relative importance of the landscape attributes in the experiment (Carlsson et al., 2007). Table 12 shows that the ranking of landscape attributes are identical in both models. We do observe substantial differences in the relative importance of landscape attributes. Following Carlsson et al., (2007) we analyze these differences by comparing the MRS of the attribute levels and, subsequently, by applying the complete combinatorial test developed by Poe et al. (2005). The choice of the numeraire attribute to calculate the MRS is arbitrary. We have chosen a high prevalence of hedges and tree lines as the numeraire, since it is the most important attribute in both models. The Poe tests show that five out of the six MRS differences are statistically significant at a critical significance level of at least 5%. Especially for the presence of livestock and for a high maize-grassland ratio, differences between the MRSs are substantial. The presence of livestock, for example, is valued at 0.59*high hedges and tree lines in the non-monetary experiment, whereas in the monetary experiment presence of livestock is valued at 0.41*high hedges and tree lines. In conclusion, although the ranks of the attributes do not vary between the monetary and non-monetary experiment, the marginal rates of substitution between attributes and attribute levels vary widely, confirming that the two experiments lead to different insights on the relative importance of the attributes and their levels.

Table 12. MNL attribute coefficients, attribute ranks and marginal rates of substitution (MRS) of the non-monetary and monetary experiment

Attribute	**Non-monetary experiment** Coefficient	Rank	MRS	**Monetary experiment** Coefficient	Rank	MRS	**p-value combinatorial test**
Presence of livestock	1.245***	3	0.59	.712***	3	0.41	0.011
Med maize-grassland	.300***	6	0.14	.171**	6	0.10	0.239
High maize-grassland	.612***	5	0.29	.172*	5	0.10	0.001
Med hedges and tree lines	1.595***	2	0.77	1.472***	2	0.86	0.045
High hedges and tree lines	2.063***	1	1	1.721***	1	1	-
Med forest patches	.215**	7	0.10	-.020	7	-0.01	0.026
High forest patches	.713***	4	0.33	.405***	4	0.24	0.040
Payment vehicle	-			-.139***			
Log-likelihood	-1,199			-1,571			
Adjusted-R^2	0.28			0.23			
N observations	1,528			1,872			
N individuals	191			234			

*** p < 0.001, ** p < 0.01, * p < 0.05

Figure 15 shows a graphical visualization of the relative importance of landscape attributes in the two models by indicating each attributes' normalized coefficient. Normalized coefficients are obtained through:

$$\beta_{i,norm} = \frac{\beta_i}{\sum_{j=1}^{J} \beta_j} \tag{2}$$

where the normalized coefficient $\beta_{i,\ norm}$ is equivalent to coefficient β of attribute i, divided by the sum of all landscape attribute coefficients j = 1,…,J. Figure 15 shows that a medium and high prevalence of linear elements are relatively more important in the experiment with payment vehicle. Other landscape attributes, especially the presence of livestock and a high maize-grassland ratio, are relatively important in the experiment without payment vehicle. The results indicate that respondents make different trade-offs and choices when a financial trade-off is introduced to the experiment. For instance, the relative importance of high levels of hedges, tree lines and maize-grassland ratio compared to medium levels of these attributes, is lower in the experiment with payment vehicle. This indicates a greater satisfaction with a medium level of these attributes when a financial trade-off is added to the experiment.

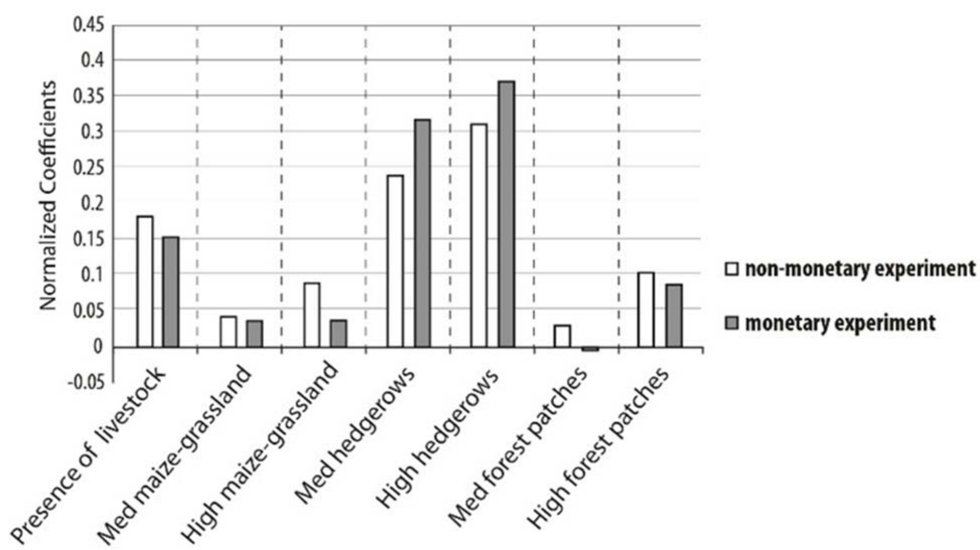

Figure 15. The relative magnitude of preferences for landscape attributes indicated by normalized coefficients estimated for the non-monetary and the monetary experiment

Effects of price-attribute non-attendance on landscape preferences

After completion of the choice tasks, respondents were asked which of the attributes they included in making their choices. Multiple boxes could be ticked in this question: the four landscape attributes and the price-attribute (only in the monetary experiment). Out of the 234 respondents who participated in the monetary experiment, 84 individuals indicated that they did not take into account the price in their choice process. This group represents the stated non-attendance sub-sample in our analysis. But how reliable is stated non-attendance for the price attribute? Figure 16 shows the adjusted R^2 values of a set of models we estimated with and without a price-attribute coefficient using the non-attendance sample, the attendance sample and the full sample of the monetary experiment. The R^2 values on the left hand side of the x-axis are obtained from models that include only the landscape attributes as explanatory variables, whereas R^2 values on the right hand side of the x-axis are obtained from models that include both the landscape attributes and the price attribute. There are three interesting observations. First, explanatory power of landscape attributes is very similar in the three samples, i.e., there are virtually no differences in the explanatory power of landscape attributes on choice behaviour between people who attend to the price attribute and people who do not. Second, the information provided by the respondents on their (non-)attendance to the price attribute corresponds very well with the patterns in explanatory power of the models. For the stated non-attendance sub-sample, the payment vehicle has no effect on the adjusted R^2, whereas for the stated attendance sample the adjusted R^2 increases from 0.17 to 0.29 when the payment vehicle is added as an explanatory variable. Third, explanatory power for the full sample is a weighted average of explanatory powers of the stated attendance and stated non-attendance samples. Although this is true by definition, the figure clearly shows that explanatory power of the full sample model is well below that of the stated attendance sample, simply because of the fact that a substantial part of respondents chose to ignore the price attribute in making their choices. We will show the consequences for monetary value estimates later on in this section.

Returning to our main research question, we explore the effect of price-attribute (non-) attendance on relative preferences for the landscape attributes and their levels. Figure 17 shows normalized coefficients of landscape attributes for the non-monetary experiment (white bars) and for the full sample, the attendance sample and the non-attendance sample of the monetary experiment (grey bars). The normalized coefficients indicate the relative magnitude of the effects, i.e., relative preferences for the landscape attribute levels. The results largely confirm our hypothesis that relative preferences for the non-attendance group are similar to the relative preferences for the non-monetary experiment. For the presence of livestock, a high maize-grassland ratio, and the prevalence of hedgerows and tree lines, relative preferences derived from the non-monetary experiment are similar to the stated non-attendance group, while they are very different from the stated attendance group. In the non-monetary and stated non-attendance groups, we observe relatively high preferences for livestock and a high maize-grassland ratio, whereas the attendance group primarily focuses on the prevalence of hedgerows and tree lines. We observe that respondents who take into account the financial trade-off, prioritize and systematically choose for the landscape attribute that they, on average, prefer most: the prevalence hedgerows and tree lines. Respondents who include the monetary attribute in their choices express high preferences for hedgerows and tree lines, and they express lower preferences for the presence of livestock and the other attributes. In comparison, respondents who do not make a financial trade-off also express relatively high preferences for the presence of livestock and other attributes in the experiment. However, the prevalence of forest patches forms a notable exception to this rule. Relative preferences for a high prevalence of forest patches are low in both the non-monetary sample and the monetary stated attendance sample.

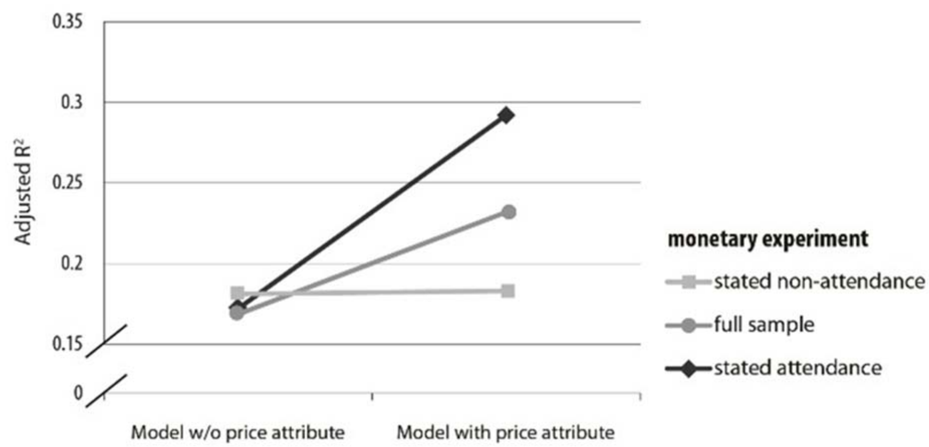

Figure 16. Adjusted R^2 values of MNL model estimates of the (non-)attendance sub-samples and the full sample of the monetary experiment with and without the price-attribute as explanatory variable

Another interesting finding is that differences in the trade-offs between non-monetary attributes between the stated non-attendance and the full sample are substantially larger than differences between the stated attendance and the full sample. In this particular case these differences do not affect full sample trade-offs to a large extent, i.e., differences in normalized coefficients between the attendance and the full sample are relatively small. This is likely the result of the fact that the share of respondents that do not include the price attribute in their choices is relatively small. However, in case studies where non-attendance to the payment vehicle is large(r) than in our case study, results for the full sample of respondents may be more substantially affected.

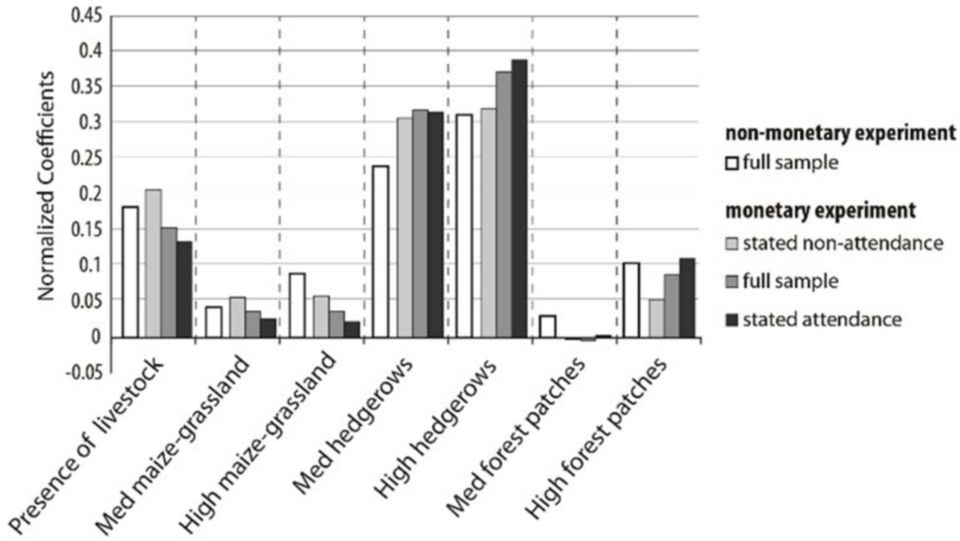

Figure 17. Normalized coefficients indicating the relative magnitude of the effects for the non-monetary experiment and MNL estimates based on the (non-) attendance sub-samples and the full sample of the monetary experiment

A combinatorial test (Poe et al., 2005) substantiates our preliminary findings. Table 13 shows the results of the Poe tests, which indicate that, in contrast to the overall results presented in Table 12, there is no significant difference between the MRS of the maize-grassland ratio and the presence of livestock in the non-monetary experiment and the stated non-attendance group. In comparison, the combinatorial test between the non-monetary samples and the stated attendance group confirms that the relative preferences for livestock and maize-grassland ratio differ significantly. However, for a high prevalence of forest patches as well as a medium prevalence of hedgerows, there are significant differences between the non-monetary and stated non-attendance group, which does not confirm our hypothesis.

Table 13. P-values from Poe's combinatorial tests on MRS between attribute levels (numeraire coefficient is 'high hedgerows' for all MRSs)

Sub-samples	Presence of livestock	Med maize-grassland	High maize-grassland	Med hedgerow	Med forest patches	High forest patches
Non-monetary vs. Stated attendance	0.001	0.107	0.000	0.222	0.067	0.210
Non-monetary vs. Stated non-attendance	0.273	0.278	0.130	0.009	0.158	0.033

4.3.4 Discussion

The empirical analysis presented in this paper provides valuable insights into differences between monetary and non-monetary valuation using choice experiments. Moreover, it presents the effects of (controlling for) non-attendance to the payment vehicle in stated preference studies.

This study addresses the question whether or not relative preference for landscape attributes change as a result of adding a payment vehicle to the choice experiment. For this we perform two choice experiments: one experiment without a price attribute, and one experiment with a price attribute. We find that the ranking of attributes in the monetary and non-monetary experiments are identical. However, the comparison of the MRS of the attributes and their levels revealed clear differences between the relative preferences for landscape attributes in our case study area. Poe's combinatorial test (Poe et al., 2005) indicated that almost all differences in MRS estimates between the two experiments were statistically significant.

We have furthermore assessed the effects of non-attendance to the payment vehicle on relative preferences for landscape attributes. We have identified sub-samples of stated price attribute attendance and non-attendance. An important finding of this study is that in our sample stated non-attendance seems to be a good indicator of actual non-attendance. We hypothesized that respondents in the stated non-attendance sample, *ceteris paribus*, have a similar preference pattern as respondents in the non-monetary experiment. Regarding both the normalized coefficients and the MRS, we can conclude that this hypothesis is confirmed for most attributes. However, for a medium prevalence of hedgerows and a high prevalence of forest patches, the hypothesis is not confirmed. Especially for a high prevalence of forest patches, the pattern of normalized coefficients in the (non-) attendance sub-samples is the opposite of what was expected.

Incorporating a payment vehicle in a choice experiment has the advantage that it provides a common denominator for preferences for the individual landscape attributes and their attribute levels. Moreover, as shown by our results, it generates more pronounced trade-offs by forcing respondents to include resource scarcity in their choices (at least for those respondents that actually include the payment vehicle in making their choices). However, we show that a substantial disadvantage of including a financial trade-off may be that a substantial share of respondents (in our case about 35%, but in some studies up to 80%, e.g., Scarpa et al., 2009) ignores the payment vehicle. The consequence is an upward bias in monetary value estimates. Ignoring the payment vehicle either implies that people really do not care about the payment, or that people choose to ignore the monetary attribute in the choice experiment (e.g., for reasons of cognitive overload or because they reject the notion of paying for landscape improvements through an increase in prices) even though in reality they would not. We argue that the former explanation is often far less likely because choice experiments generally include monetary changes of substantial magnitude. In our case the monetary attribute has relatively low absolute value. However, in case it is implemented through tourist taxes, the values would lead to a price increases in the range of 7% to 70% (depending on the price attribute level and varying initial prices). These observations are related to the argument that attribute non-attendance may be confounded with preference heterogeneity (Hess et al., 2013). We argue that this confounding likely does not hold in our case, for two reasons. First, given the magnitudes of price increases included in the choice experiment, it is highly unlikely that respondents do not care about these changes in reality. Second, respondents actually state that they did not at all include the price attribute in making their choices, and MNL models reveal that the price coefficient for this sample is indeed very small and statistically insignificant. These observations suggest that payment vehicle non-attendance, at least in our case, represents a form of hypothetical bias which can and should be corrected for.

An implication of our results for landscape and cultural ecosystem services research is that monetary and non-monetary preference estimates thus require different interpretations. When studying social-cultural values to estimate how a public good contributes to ecosystem services and well-being, a non-monetary choice experiment provides sufficient insight in the relative preferences between different landscape attributes. In comparison to other landscape preference studies the respondents have to make trade-offs between landscape attributes, leading to a clear elicitation of the preferences. If a study is used to develop a tourist tax or to provide input for a cost-benefit analysis, one obviously needs a monetary experiment. Still, the success rate of avoiding hypothetical bias in monetary experiments heavily depends on the design and the focus of the experiment. For some more intangible public goods, such as landscape aesthetics, a credible relation between a payment vehicle and the attributes of the good itself is difficult to establish (Turner et al., 2010). Those experiments could encounter unacceptably high levels of payment vehicle non-attendance and, therefore, yield severely biased WTP-estimates. In those cases, controlling for stated non-attendance to the payment vehicle is highly recommended to obtain accurate monetary value estimates.

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4.4 Regret minimization as an alternative to utility maximization[13]

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4.4.1 Introduction

Consumer preferences are a critical factor in the development of successful Alternative Fuel Vehicles, or from here on AFVs (e.g., Struben and Sterman, 2008; Huijts et al., 2012). For this reason a wide range of recent studies have sought to explore these preferences in the context of a range of AFV-technologies (e.g., Erdem et al., 2010; Ozaki and Sevastyanova, 2011; Hoen and Koetse, 2014; Koetse and Hoen, 2014). As a recent overview-study by Roche et al. (2010) suggests, many studies into consumer preferences for AFVs rely on the estimation and subsequent application of discrete choice models on stated choice-data. Moreover, what is also shared by the large majority of these and other studies into consumer preferences for AFVs is that they adopt a particular behavioural model for the analysis of observed choices: that of utility maximization. More particularly, almost without exception estimated discrete choice models take on the form of so-called Random Utility Maximization (RUM) models with linear-in-parameters utility functions (see Ben-Akiva and Lerman, 1985, and Train, 2009, for in-depth discussions of the RUM-model of consumer preferences).

Notwithstanding the obvious elegance and tractability of these models as exhibited in a wide range of studies, the almost exclusive focus on RUM as a model of behaviour is not in line with recent trends in adjacent fields, where non-RUM models have gained popularity lately as possibly more behaviourally realistic alternatives to RUM (e.g., Arentze and Timmermans, 2007; Hensher, 2010). A micro-simulation study by Mueller and de Haan (2009) also shows how capturing so-called bounded rational behaviour may lead to new insights into consumer preferences for AFVs. Although this latter study, like most others in the field, uses a linear-in-parameters RUM-based decision-rule, it does allow for insights from non-utilitarian behavioural models (such as Prospect Theory) to co-determine consumer preferences and choices.

[13] This section is based on: Chorus CG, Koetse MJ, Hoen A, 2013, Consumer Preferences for Alternative Fuel Vehicles: Comparing a Utility Maximization and a Regret Minimization Model, *Energy Policy* **61**, 901–908.

Motivated by this recent interest in non-RUM decision-rules, this paper proposes to use a so-called Random Regret Minimization-based behavioural model that has recently been successfully introduced in a range of travel demand studies. This so-called RRM-model (Chorus, 2010) postulates that consumers aim to minimize regret, rather than maximize utility, when making decisions. The RRM-model (its Multinomial Logit-form) distinguishes itself from other non-RUM models in terms of its usability: it features closed-form formulations of choice-probabilities and can be easily estimated using readily available discrete choice-software. The model has been successfully tested empirically by various researchers in the context of a wide range of travel demand related choice-contexts, but also in the context of choices made by politicians among policy options and choices made by visitors of dating websites among dating profiles (see Chorus, 2012, for an overview of the empirical evidence). The model is based on the behavioural notion that regret emerges when a non-chosen alternative outperforms a chosen one in terms of one or more features. It should be noted here that it is well known in the field of consumer research (e.g., Zeelenberg and Pieters, 2007) that the minimization of regret is a particularly important determinant of consumer behaviour when choices are perceived by the decision-maker as important and difficult, and relevant to his or her social peers. Furthermore, the regret minimization model in a conceptual sense puts extra ‘weight’ on situations where a considered alternative performs relatively poorly compared to the competition.

The RRM-based model postulates that, when choosing between alternatives, decision-makers aim to minimize anticipated random regret. Systematic regret is conceived to be the sum of all so-called binary regrets that are associated with bilaterally comparing the considered alternative with each of the other alternatives in the choice set. The level of binary regret associated with comparing the considered alternative with another alternative *j* is conceived to be the sum of the regrets that are associated with comparing the two alternatives in terms of each of their *M* attributes. Aside from their similarities (such as Logit-type choice probabilities), the RUM and RRM modelling perspectives exhibit a number of important differences; see Chorus (2012) for an in-depth discussion of these differences. An important difference is that the model based on regret minimization implies a particular type of semi-compensatory behaviour. This is a direct result of the form of the regret-function. Improving an alternative in terms of an attribute on which it already performs well relative to other alternatives generates only small decreases in regret. In contrast, when the performance on another equally important attribute, on which the alternative has a *poor* performance relative to other alternatives, deteriorates to a similar extent, this may generate substantial increases in regret. As a result, the extent to which a strong performance on one attribute can make up for a poor performance on another depends on the relative position of each alternative in the set. This results in the so-called compromise-effect which has been well established empirically in the field of consumer choice (e.g., Simonson, 1989; Wernerfelt, 1995; Kivetz et al., 2004) while being ignored by linear-additive RUM-models. This effect states that alternatives with an ‘in-between’ performance on all attributes, relative to the other alternatives in the choice set, are generally favoured by choice-makers over alternatives with a poor performance on some attributes and a strong performance on others. In section 5, we highlight how the fact that the RRM-model captures a compromise effect while its linear-additive RUM-counterpart does not, leads to non-trivial differences in choice probability simulations and policy implications in the context of our data.

4.4.2 Data collection and model estimation

For assessing the differences between utility maximization and regret minimization models, we use a choice experiment among lease car drivers in The Netherlands. The choice experiment uses respondents from the automotive panel of TNS-NIPO. This panel contains more than 40,000 households with one or more cars, approximately 4,000 of which are company car drivers. The panel is established through random sampling, meaning that each member of society has an equal chance to be added to the panel as long as he or she conveys the willingness to cooperate. Each respondent was presented with eight choice tasks consisting of three options each, and was asked to indicate his or her preferred option. We generated a so-called efficient statistical design using the Sawtooth CBC software package, which was also used to produce and field the online questionnaire. The design consisted of 30 survey versions of 8 choice tasks each. The final version of the questionnaire was fielded in June and July of 2011. In total we obtained 616 complete questionnaires, leading to 4,927 usable observations (after deletion of one missing observation).

The attributes used in the choice experiment were selected based on consultations with stakeholders and a literature review. An important criterion for selection was that there was a marked difference between current cars and some or all AFVs. Another criterion was that the attribute is considered to be crucial for car choice, both from an expert opinion point of view as well as from the literature. We first included car type (i.e., type of AFV – electric, fuel cell etc.) as an attribute, simply because we also want to get insight into preferences for AFV's apart from their attributes. We included eight other attributes, i.e., purchase or catalogue price, monthly contribution (this attribute is explained further below), tax percentage charge, driving range, recharge/refuelling time, additional detour time for refuelling, number of available brands/models, and policy measure. For details on the attributes and their levels we refer to Chorus et al. (2013).

We use a linear model specification and models were estimated using the freeware discrete choice-package Biogeme (Bierlaire, 2003; 2008). See Table 14 for estimation results. To start with the comparison of the RUM- and RRM-models, observe that – as expected – all parameters have the same sign in both models, and that significance levels (as implied by the *t*-values) and *relative* magnitude of parameter estimates are also quite similar across models.

Table 14. Estimation results for RUM- and RRM-models

	RUM		**RRM**	
Attribute/Variable	*Beta*	*t-value*	*Beta*	*t-value*
Constant Fuel cell	-.383	-2.58	-.396	-2.74
Constant Electric	-1.16	-6.67	-1.16	-6.71
Constant Flexifuel	-.009	-.07	-.0306	-.22
Constant Hybrid	.018	.13	.00137	.01
Constant Plug-in hybrid	-.693	-5.13	-.709	-5.40
Access to bus lanes	.0738	1.40	.0737	1.39
Free parking	.0939	1.81	.0915	1.77
Purchase price	-.0000231	-4.46	-.0000154	-4.60
Tax percentage charge	-.0304	-9.67	-.0203	-9.68
Personal monthly contribution	-.00242	-18.72	-.00159	-18.81
Driving range	.00137	8.55	.000954	8.64
Recharge/refueling time	-.00119	-4.31	-.000757	-4.36
Additional detour time for refueling	-.0172	-7.05	-.0112	-7.21
Number of available models	.00112	4.48	.000748	4.47
Final LogLikelihood	*-4,193*		*-4,192*	
Rho-square	*.225*		*.225*	

Differences in model fit between the RUM- and the RRM-specification are also very small, although it is found that the RRM-model achieves a final log-likelihood that is slightly higher than that of the RUM-model, signalling a slightly better fit with the data (note that initial LogLikelihoods are the same for the two models). This difference of 1 LogLikelihoodpoint is very small and only significant at a 10%-level when put to the Ben-Akiva and Swait (1986) test, and is as such trivial from a practical viewpoint.

Note that while an overview study (Chorus, 2012) finds that in general differences in fit between RRM- and RUM-models are marginal, the difference obtained in the context of our data is exceptionally small. Also note that the difference is much smaller than the difference obtained in Hensher et al. (2011) in the context of their stated choice data concerning preferences for AFVs.

4.4.3 Model validation

More importantly perhaps than differences in model fit are differences in out-of-sample predictive ability. This section provides a validation exercise in which we focus on out-of-sample predictive ability of the two estimated models. More specifically, the sample was split into two parts, an estimation sample containing roughly two-thirds of cases, and a validation sample containing the remaining roughly one-third of cases. To be precise, each case was randomly assigned a value from a uniform distribution between 0 and 1, and cases with values lower than 0.666666… were assigned to the estimation sample; the remaining ones being assigned to the validation sample. This process was performed twice, so that two replications of the validation exercise were obtained. Estimation results of the two models (RUM and RRM) on the estimation sample are very similar to those reported in Table 14 (in terms of parameter values as well as model fit statistics) and are not reported here for reasons of space limitations. Two types of validation exercises are performed in the context of the validation sub-sample and are summarized in Table 15. First, the mean-likelihood of chosen alternatives is computed for both models. More specifically, we re-estimated the models without those parameters that obtained a robust p-value lower than 0.15, and used the results from this re-estimation to simulate choice probabilities for cases in the validation sample. That is, for each choice task in the validation sample the predicted choice probability is computed for the chosen alternative, after which the mean of these probabilities is computed.

Table 15. Validation results for RUM- and RRM-models

	RUM	**RRM**
Mean-likelihood (replication 1)	.492	.497
Mean-likelihood (replication 2)	.501	.505
Mean-likelihood (average)	*.497*	***.501***
Mean hit rate (replication 1)	.621	.617
Mean hit rate (replication 2)	.639	.632
Mean hit rate (average)	***.630***	*.624*

A high mean-likelihood suggests that the model estimated on the estimation sample does well in the sense that it assigns high choice probabilities to the chosen alternatives in the validation sample. It appears that, in terms of this metric, the RRM-model slightly outperforms its RUM-counterpart. Results are very similar across the two replications. As a second way to test the models' predictive ability, so-called hit rates are computed for each model by comparing, again for each choice task in the validation sample, the alternative that has the highest predicted choice probability with the actually chosen alternative. If the predicted and the chosen alternative are the same, a score of one is given to that case, and zero otherwise. The average of these values across observations is called the hit rate. A high hit rate suggests that the model estimated on the estimation sample performs well in terms of identifying the most attractive alternative for choice situations in the validation sample. Results suggest that the ability of the RUM-model to correctly identify the most attractive alternative in a set is slightly higher than that of the RRM-model, in the context of our data. Again, results are very similar across the two replications.

In sum, when it comes to out-of-sample predictive ability, there appears to be no clear 'winner' among the two models. Combined with the very small differences in terms of model fit, this suggests that rather than applying one of the two models for choice probability simulations and policy implication derivations, it would be more sensible to use both models and compare results. This is done in the next section.

However, before we do this, we wish to point out that, although differences between the RUM- and RRM-model in terms of model fit and mean predictive ability are very small, the two models can indeed generate non-trivial differences in terms of predicted choice probabilities for specific choice situations. We show this by comparing across the two model types, for each case in the two validation samples, the computed choice probability for each of the three alternatives in the choice set. More specifically, we compute and report in histogram-form the distribution of absolute values of the differences (in percentage points) between the RUM-probability and the RRM-probability. Figure 18 shows the results.

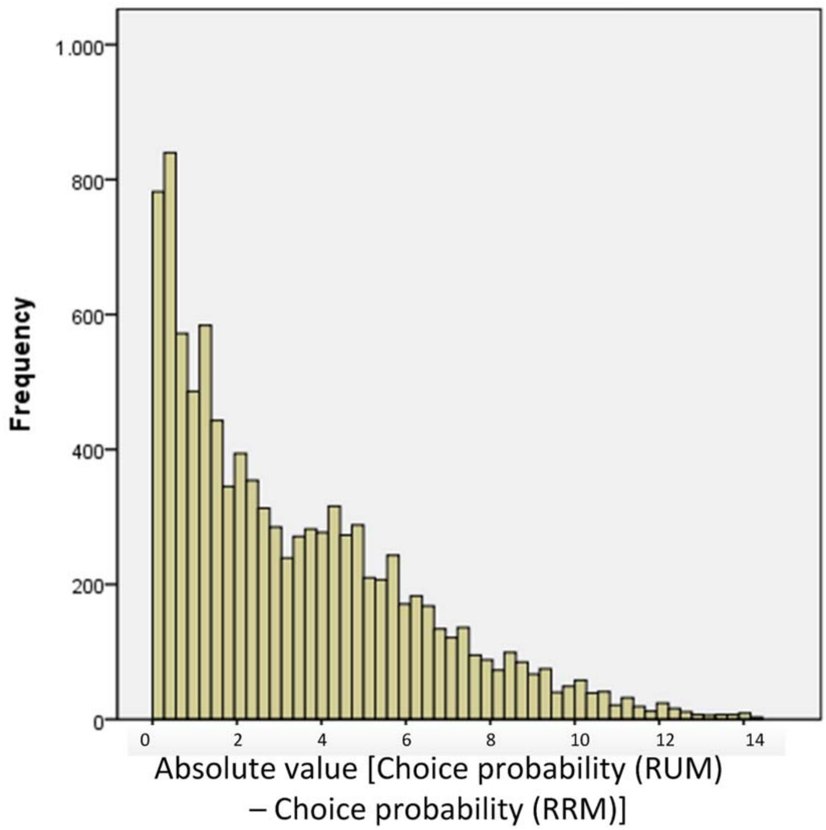

Figure 18. Differences in predicted choice probabilities for AFVs (percentage points)

As the figure shows the two models generate very small differences in choice probabilities for the majority of choice sets in the validation sample. However, for a substantial number of choice sets the difference in choice probabilities is more than five percentage points, and for a non-trivial number of choice sets the difference is ten percentage points or more. Furthermore, a finding that is not reported in Figure 18 is that we find that in no less than 7% of all cases, the two models identify different alternatives as the most popular ones ('winners') in particular choice tasks. This result highlights that, although aggregate statistics of model fit and predictive ability may not differ much between RRM- and RUM-models, the two model types may in fact generate markedly different choice probabilities (and: policy implications) at the level of individual choice sets. In the next section, we explore two such individual choice sets in more detail.

4.4.4 Choice probability simulations

As has been argued in section 2, one of the main differences between the RUM-model and its regret-based counterpart is that the latter exhibit's the compromise effect, which states that a choice alternative receives a choice probability bonus when it 'moves to the centre of the choice set', figuratively speaking. In this section we explore how this conceptual difference between the two models plays out empirically in the context of the two models estimated on our data. For this we focus on a choice set which, although fictive, may be considered a realistic result of AFV-technology developments in the medium term future (see Table 16). We explicitly position vehicle C (fuel cell) as a compromise alternative in that it is positioned in between the other two vehicles in terms of six (out of 7) relevant attributes. Note that it would be unrealistic to assume values for the remaining attribute (extra detour for refuelling) such that the fuel cell-vehicle would be a 'full' compromise alternative. Parameter estimates shown in Table 14 are used for simulating choice probabilities for the three vehicle types. This is done by applying the logit-equations for the RUM- and RRM-models, while using as input the estimated parameters as presented in Table 14 and the attribute levels as specified in Table 16.

Table 16. Choice probability simulations for AFVs (fictive example based on possible medium term-future in terms of vehicle technology)

	Vehicle A	**Vehicle B**	**Vehicle C (compromise)**
Technology	Electric	Conventional	Fuel cell
Purchase price	40,000 euro	25,000 euro	35,000 euro
Tax percentage charge	0 %	14 %	7 %
Personal monthly contribution	400 euro	300 euro	350 euro
Driving range	75 km	500 km	250 km
Recharge/refueling time	480 minutes	0 minutes	25 minutes
Additional detour time for refueling	0 minutes	0 minutes	30 minutes
Number of models	1	500	5
Choice probability (RUM)	*3.7 %*	***85.6 %***	*10.7 %*
Choice probability (RRM)	*3.6 %*	***83.5 %***	*12.9 %*

As shown in the two rows at the bottom of the table the RUM-model assigns a lower choice probability ('market share') to the compromise vehicle than does the RRM-model, the difference amounting to slightly more than 2 percentage points. In other words, the RRM-model, by featuring a compromise effect, predicts a choice probability for the fuel cell vehicle that is around 20 percent higher than the share predicted by the RUM-model. While this difference is small in light of the very large choice probability of the conventional technology-vehicle (B), it is non-trivial from the viewpoint of AFV-technology developers.

Based on this analysis the following conclusions seem warranted. First, in line with theoretical expectations, the RRM-model assigns higher choice probabilities to so-called compromise alternatives than the RUM-model. Second, resulting differences in predicted choice probabilities between RRM and RUM are of a non-trivial magnitude. Third, in some situations (where alternatives are close competitors in terms of their popularity), a situation may occur where the two models identify different 'winners' in the choice set, potentially leading to markedly different policy implications.

4.4.5 Conclusions

This paper investigates consumer preferences for Alternative Fuel Vehicles (AFVs) based on a stated choice-experiment among Dutch company car-leasers. We estimate not only a conventional linear-additive Random Utility Maximization-model but also a model that is based on the notion that choices for durable goods like vehicles are often driven by a consumer's wish to avoid regretting the choice ex post.

Random Regret Minimization-models were compared with their RUM-counterparts, and a number of relevant new findings are obtained. First, it appears that the two models perform almost equally well in terms of model fit, the RRM-model having a very slight edge over its RUM-counterpart. In terms of predictive ability on a validation sample, differences between the two models are also small, yet somewhat larger than the differences in model fit. In terms of the mean predicted likelihood of chosen alternatives, the RRM-model does slightly better than RUM. The opposite is the case when it comes to correctly predicting the most popular alternative in a set (the so-called hit rate). Analyses show that in the validation sample, differences in terms of predicted choice probabilities for alternative vehicles are on average more than 3 percentage points. For a non-trivial share of choice tasks, differences are larger than 10 percentage points. In more than 7% of cases, the implied most popular alternative in the choice set is different for the RRM model than for the RUM model.

Choice probability simulations in the context of fictive choice sets highlight an important property of RRM-models. The RRM-model assigns substantially higher choice probabilities to so-called compromise alternatives (i.e., alternatives with a reasonable, rather than extreme performance on relevant attributes) than its RUM-counterpart. As such, the RRM-model results in a policy-implication that it may be welfare enhancing to provide an alternative that is 'in the centre of the choice set', and as such make it a compromise alternative. More generally, the RRM-model provides a behavioural base for quantitative analyses of the impact of so-called choice set-engineering policies. These policies aim to influence market shares of a product or service by focusing not only on (improving) that product's own characteristics, but by also paying explicit attention to the positioning of the product – in terms of relevant attributes – relative to the other alternatives available in the choice set.

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5 An integrated ecosystem service assessment model

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5.1 Introduction

In a world of scarce resources and limited opportunities, decisions made without economic analyses which consider the natural environment are likely to deliver poor value for money. Simply ‘letting the market decide’ the worthiness of different policy options is insufficient, as a narrow focus on market interests ignores the myriad non-market impacts of environmental change (Bateman et al., 2013; Pearce, 1998; Pearce et al., 1989). Using an integrated modelling approach to include the economic value of non-market goods has potential to make use of scientific evidence to significantly improve the social value of public spending. Veritably, many of the services provided by the natural environment can be assessed using economic values which can be then readily incorporated within decision making systems (Bateman et al., 2011b; Champ et al., 2003; Heal et al., 2005; Pearce, 1998). At its most basic level, this report addresses one simple question: “What is the best use of land?”

As is perhaps predictable, this seemingly innocuous question returns a series of complex answers that differ substantially depending on the comprehensiveness of the analyses undertaken. However, simply using and understanding clear and consistent terminology – particularly around ‘market prices,’ ‘social values,’ and ‘economic values’ – can address much of this complexity. Economic value is a commonly misused term. Also known as ‘social value’ (Bateman et al., 2014) it incorporates the full contribution of any good or service to human welfare, and is composed of both market and non-market elements. For instance, the economic value of a change in forest cover may include market impacts from timber production as well as non-market impacts from carbon sequestration, open access recreation, nutrient cycling, changes in water quantity and quality, air purification and maintaining habitats for wild species.

Market prices (aka exchange prices) reflect the interaction of supply and demand for goods and services traded in formal markets. By definition, they ignore externalities (non-market impacts) such as unregulated environmental degradation. Clearly, confusing the subset of market prices with broader economic values yields only a partial and ultimately misleading conception of environmental-economic interactions. The ecosystem services approach offers a useful lens through which to view and understand the myriad market and non-market sources of value generated by the natural environment.

5.2 Ecosystem service decision support tools: a review

Amid growing recognition of the natural environment's role in generating human wellbeing, a series of ecosystem service decision support tools have been developed to guide decision making. These vary in sophistication from simple spreadsheet tools to complex software packages integrating biophysical, GIS and economic models (Bagstad et al., 2013) and draw upon many fields, including ecology, hydrology, geography, systems theory, economics and the social sciences. They also differ in their ability to value changes in ecosystem services and handle various spatial and temporal scales, data and computational constraints and conflicts between users, science and data (van Delden et al, 2011).

A new class of *integrated* ecosystem service mapping tool including InVEST, LUCI, MIMES and The Integrated Model (TIM, outlined in detail below) is beginning to emerge. These tools incorporate state-of-the-science biophysical models to reflect interactions between multiple ecosystem services at various spatial and temporal scales. Their process-based biophysical underpinnings enable these tools to use information from areas with high data availability to model environmental processes and relationships in areas where data is relatively scarce (Bateman et al., 2011a). This greatly enhances coverage, and thus the likelihood that a given tool can be applied to specific policy questions.

Two of the best known ecosystem service tools are InVEST (Tallis et al., 2013) and ARIES (Bagstad et al., 2011). InVEST currently considers water quality, soil conservation, carbon sequestration, biodiversity conservation, aesthetic quality, coastal and marine environment vulnerability, hydropower production, pollination services and values of selected market commodities. Its models are biophysical, and include explicit economic valuation of all services. The most recent release of ARIES includes carbon sequestration, flood regulation, water supply, sediment regulation, fisheries, recreation, aesthetic viewsheds, and open-space proximity value. It is designed to be extremely flexible and can include biophysical models where desired, but generally uses empirical statistical approaches to extract relationships between inputs and outputs.

Other tools gaining interest are MIMES, LUCI and Co$ting Nature. For a review, see Bagstad et al. (2013). MIMES is a systems model which represents the dynamics and feedback loops between physical, social and economic processes. It seeks to be a truly integrated model, and represents an ambitious effort to take integrated modelling forward to match or extend the state of the art of meteorological and climate modelling to economic models. LUCI is highly spatially explicit (with resolution of 5 meter grid squares within the UK and at worst 50 by 50m globally) and may therefore be applied at any spatial scale, say for considering the cumulative impacts of small interventions such as riparian planting at national scale. It currently considers agricultural productivity, flood regulation, carbon sequestration, sediment regulation, habitat connectivity, and water quality. It has a simple approach to considering trade-offs between services, classifying individual service provision at its native spatial resolution into "existing good", "potential to improve", or negligible existing or potential provision". It then layers those categorised services to identify parts of the landscape where trade-offs versus win-win situations exist, and where management interventions could enhance or protect multiple services. Finally, Co$ting Nature uses global datasets to estimate and value water yield, carbon storage, nature-based tourism, and natural hazard mitigation services, aggregating these into a "service index" accounting for not only provision but also beneficiary location. Although it is less flexible and modular than the other frameworks, it is significantly easier to apply (and access). Table 17 offers a brief comparison of The Integrated Model (TIM) against three common ecosystem service decision support tools. A more detailed comparison may be found in Bateman et al. (2014) and Bagstad et al. (2013).

TIM is the first application of an integrated modular ecosystem service framework covering the whole of the UK and using detailed UK-specific data (we discount "global" applications, using coarser global data, such as MIMES (Boumans et al., 2015) and Co$ting Nature). Compared to the established suite of ecosystem service models, TIM's novelty lies in the introduction of formal optimisation alongside ecosystem service valuation. Crucially, because services are valued in common economic units, trade-offs and comparisons can be drawn and their impacts can be readily interpreted by a diverse audience of varying specialist backgrounds. This is particularly useful in land use policy as decision makers are expected to maximise net benefits derived from the scarce resources at their disposal, accounting for a broad range of biophysical and economic impacts and responses. Although InVEST also applies economic valuation, it stops short of formal optimisation and lacks the rich, custom data set at the 2km grid square resolution used by TIM in the UK.

Table 17. Overview of integrated ecosystem service modelling frameworks (source: Bateman et al., 2014).

	ARIES	Co$ting Nature	InVEST	TIM
Model approach	Bayesian belief network and agent based modelling; flexible framework	Web-enabled model with globally available data using simple empirical models.	Detailed biophysical models and economic valuation of all services	Biophysical modules with robust economic valuation and formal optimisation
Spatial scale of analysis [resolution of individual elements in brackets if applicable]	Flexible, but generally regional scale	Flexible, has global coverage [1km^2 or 1ha]	Regional – component models not suited for local scale application	Medium catchment to national [2km grid square]
Temporal scale of analysis	Flexible	Steady state	Annual, sub-annual in development	Annual but could be sub-annual
Data gathering effort required by user	Heavy for new applications (existing applications will be made available via web portal)	Negligible; data pre-loaded and available via web portal	Heavy	Negligible; data is pre-loaded and available within the TIM software
Parameterisation effort required by user	Theoretically low – data driven model approach; but models need to be trained on data	Negligible; default parameters provided although can be tweaked by user	Heavy –detailed biophysical models requiring parameterisation	Economic valuation of services & analysis of their inter actions
Flexibility/modularity	Very high	Low	High	High, with built-in constrained optimisation procedure
Economic valuation provided?	No	No	Yes	Yes.
Types of trade-offs considered	Biophysical & via analysis of service flow from provision to beneficiaries	Services categorised & flow to beneficiaries considered	Biophysical and monetary units traded against each other	Trade-offs analysed by explicit economic valuation of all services
Optimisation?	Through scenario optimisation; although Bayesian framework potentially enables robust optimisation and uncertainty analysis.	Through scenario optimisation only	Through scenario exploration only	Yes, constrained optimisation procedure is part of framework
Unique Features	Sophisticated modelling of flows to beneficiaries, source and sink, flexibility, Bayesian & agent based modelling.	Globally available, simple to use, data pre-loaded for user	Most established/ advanced suite of biophysical models, explicit economic valuation	Constrained optimisation procedure; explicit economic valuation; increased integration via coupling linkages between services

5.3 Scotland's ecosystem services: an integrated, modular approach

This research seeks to contribute to the efficient management of Scotland's natural resources by providing guidance for improved decision-making. The agriculture and forestry sectors play a particularly important role in shaping landscapes and ecosystems in Scotland. Land use management practices have, undoubtedly, an enormous effect on the wider delivery of many ecosystem services and related goods, such as recreation and biodiversity.

Future land use is dictated by complex interactions between economic activity (i.e. market conditions), natural environmental processes and drivers (e.g. climate change) and policy intervention (Fezzi and Bateman, 2011). While typical land use appraisals might consider a small number of pre-determined options, each described in terms of a different end-point or state, such analyses give no indication regarding which policies might be required to achieve that state. Moreover, the decision maker has no means of knowing whether the best option is included in the analysis and the chosen option may not be efficient in that it may not offer best value for money for society.

The research attempts to model the numerous physical and economic processes which characterise land, its use and the consequences of that use. These individual analyses are 'component modules' of an integrated assessment which begins with observable present-day realities and models the impacts of changes over time to yield analytically-defined end states. Monetary valuation is used as a common metric to estimate the economic value of ecosystem services and derived goods (Figure 19). Other benefits derived from nature that are vital to human well-being are quantified but not monetised.

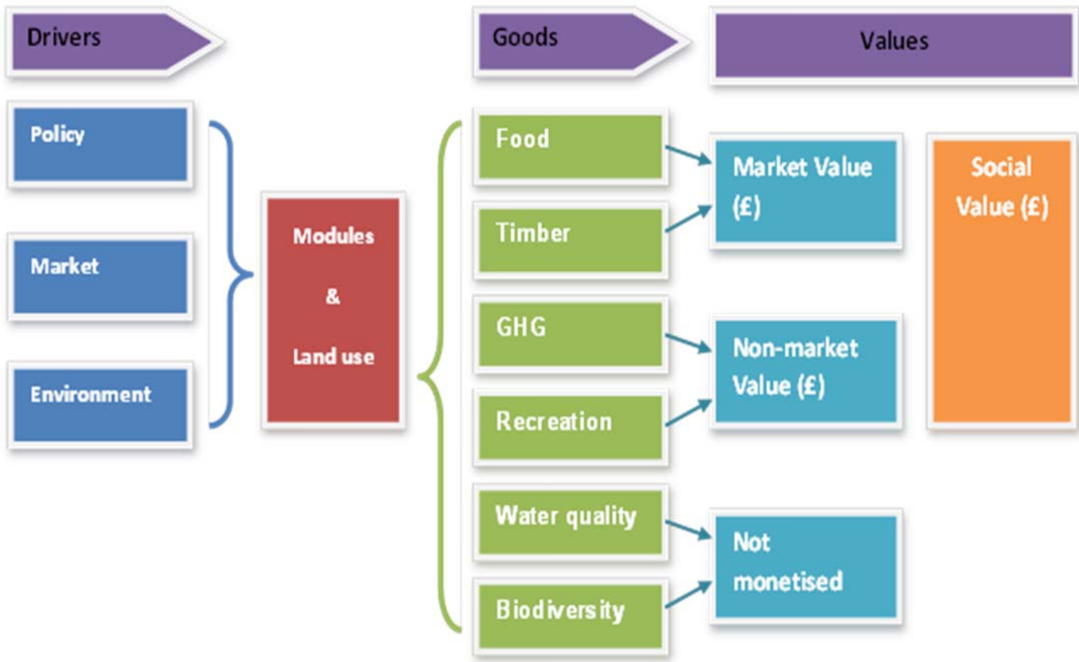

Figure 19. A schematic overview of the integrated model: multiple determinants of land use and economic value

5.3.1 Component modules

The integrated model describes multiple determinants of land use over a national extent, discretised into square grid cells at a 2 km base resolution. Temporal resolution is annual by default. Further details, including data sources, assumptions and valuation techniques on all component modules are available in (Bateman et al., 2014).

Modules to determine market values

Some ecosystem goods and services are traded in private markets and have reliable market values that must be incorporated into the analysis (e.g. food and timber). However, making decisions purely on the basis of market values (e.g. maximising gross domestic product) is unlikely to deliver optimal outcomes as it ignores non-market impacts.

Food

The agricultural production module determines land use (arable, grassland and grazing, livestock etc.) based on maximising agricultural profit as a function of soil characteristics and in response to climate trends. Adjusting these market prices for policy interventions (e.g. taxes and agricultural subsidies) enables the determination of the market value of agricultural land use.

Timber

The timber production module captures variation in growth rates, timber yield class, and timber profits for a variety of physical environmental conditions, taking into account the effects of unmitigated climate change (UK Climate Impacts Programme (UKCIP), 2009). This model predicts timber production costs and benefits for different tree species across locations.

Modules to determine non-market values

A range of non-market externalities resulting from changing land use are considered. Greenhouse gases and recreation lack market prices, but may still be valued in monetary terms using a range of non-market valuation techniques (Atkinson et al., 2012; Bateman et al., 2002; Bateman et al., 2011b; Heal et al., 2005). Values for some ecosystem services, such as water quality and biodiversity, cannot be estimated robustly.

Greenhouse Gases

There are two Greenhouse Gas (GHG) modules: farm carbon (machinery, soils, crops, fertilisers, livestock etc.) and forestry (soils and organic matter). The agricultural GHG module evaluates the carbon dioxide equivalent (in tonnes; tCO_2e) GHG flows from agricultural land use, taking into account temperature and soil characteristics. The forestry GHG module estimates GHG exchanges between the atmosphere, forest ecosystems and the wider forestry sector as a result of tree growth, mortality and harvesting. Specifically, the module incorporates the net annual carbon flows in livewood stands, harvested wood products, deadwood and forest soils, for representative conifer (Sitka Spruce) and deciduous (Pedunculate Oak) species.

There is no nationally or globally agreed value for carbon (or CO_2 equivalent) emissions. This research considers a range of carbon values based on UK government guidance (DECC, 2009, 2011; DECC Modelling, 2013) with different price paths over time, and demonstrates the sensitivity of results to alternative specifications of carbon prices. This report adopts the carbon values based on the target consistent Marginal Abatement Cost approach (non-traded sector) values (hereafter referred to as the ‘upper carbon pricing regime’). Additionally, an estimate for the Social Cost of Carbon is used for comparison (hereafter referred to as the ‘lower carbon pricing regime’).

Recreation

Impacts of changing land use on individuals’ visitation choices and associated recreational values are modelled using a random utility model (McFadden, 1974) and data on recreational visits from the Monitor of Engagement with the Natural Environment (Natural England, 2010). Impacts of substitute availability upon the number and value of visits, including the dynamic effects of progressive land use change over time, are also considered.

Water quality and biodiversity

The water quality module describes the hydrological processes that link land use to nutrient concentrations and ecological status in rivers. The biodiversity module provides a model of bird diversity related to land use. In present analyses, water quality and biodiversity are quantified but not monetised. These analyses, however, could act as external constraints e.g. a policy is implemented such to maximise society's net benefits from land use, subject to the constraint that there is no net loss in biodiversity.

Annuities

The economic assessment of land use change only makes sense if 'natural' time periods for each process are considered, the net present value of the corresponding stream of costs and benefits over those periods are calculated, and then the annualised equivalent (the 'annuity') of that discounted stream of values are calculated. This allows a fair comparison across very different activities.

5.4 Pressures and possibilities

5.4.1 Continuing climate change

Land use does not stay constant as climate change drives alterations in agricultural activities. Anticipated climate change is captured using predictions of mean growing season (April—September) air temperature and total monthly precipitation (derived from UKCP09 (2009) medium emissions scenario).

5.4.2 Shifts in policy

Currently modelled policy drivers include Single Payment Scheme (SPS; transfer payments from taxpayers to farmers) and an afforestation policy. The integrated model can opt to include or exclude SPS as a simple flat-rate average. Implementation of afforestation is presented as a case study.

5.5 A new approach to decision-making

Policy implementation considering the natural environment can significantly enhance value for money. A limited focus on, for instance, displaced agriculture alone can result in decisions which represent very poor value for the taxpayer, while a more comprehensive assessment of the wider benefits of land use change can identify new ways of applying policy which generate major gains across society. A policy-relevant case study was undertaken to examine the potential for establishing new forests in Scotland. Targeted policies deliver greatly improved value for money from available natural resources.

5.6 Constrained optimisation

At a fundamental level, economics is the science of constrained optimisation: constrained because of the scarcity of resources; and optimisation to extract the greatest possible net benefit from them. Thus defined, achieving the best use of land – extracting the greatest benefit from scarce resources – is, in its simplest distillation an economic exercise in constrained optimisation. This report argues that the value that society derives from its land is a product of the ecosystem goods and services that a) are traded in private markets and have reliable market values (e.g. food and timber) and b) lack market prices (e.g. recreation and GHG flows). Land use appraisals should reflect both of these.

5.7 Case study: planting new forests in Scotland

A policy context was established in which Scotland decides to plant 5,000 hectares of new woodland per annum for each year between 2014 and 2063, yielding an overall increase in forest extent of 250,000 hectares across Scotland over the full 50 year assessment period. This was prompted by government announcements of an intention to expand forestry in England, Scotland and Wales, and on the basis of direct discussions with the UK and devolved Parliaments (IPF, 2012; NCC, 2012; Scottish Government, 2012; Welsh Assembly, 2012).

5.7.1 Business-As-Usual baseline

Land use does not stay constant over the analysis period as climate change drives alterations in agricultural activities. A 'Business As Usual' (BAU) baseline for land use is defined against which subsequent analysis and results are compared. The baseline holds constant policy drivers, i.e., there is no afforestation, while agricultural markets (refer to drivers depicted in Figure 19) respond to changes in natural environment conditions, specifically temperature and precipitation during the growing season.

Full BAU results are provided in Bateman et al (2014). In brief, the generally warmer temperatures for Scotland will boost agricultural production and farmers will respond by moving towards more profitable activities (arable) in eastern Scottish lowlands and pushing (and intensifying) improved grassland and livestock activities further into uplands and western Scotland. Average annual emissions of CO_2e will increase as a result. Annual tree growth rates (yield) will increase marginally for representative conifer (Sitka Spruce) and deciduous (Pedunculate Oak) species. There will be small increases in bird diversity across Scotland.

5.7.2 Policy implementation strategies

The introduction of an afforestation policy will affect land use over the 50-year period from 2014 to 2063. Specifically, land is taken out of agriculture and replaced with forestry. Representative conifer (Sitka Spruce) and deciduous (Pedunculate Oak) species are modelled separately (mixed planting is not considered). The integrated model predicts response for component systems (refer to Figure 19). Although trees are only planted in Scotland, natural flows (e.g. water) and people are permitted across artificial administrative boundaries (i.e. Scotland/ England border). The pattern of planting, across space and time, which maximises the value of the measure being investigated, is identified through the integrated model. Measures, or optimisation rules, are detailed below.

Market values only

Under the ‘Market Value’ (maxMV) option the desired new afforestation is located to maximise the net benefits in terms of the market priced goods concerned (agricultural outputs and forest timber values). All non-market externalities all excluded.

Market and non-market values

Under the ‘Social Value’ (maxSV) option new forests are located to maximise the net benefits in terms of all the economic values (market and non-market values) from component modules (agricultural outputs, forest timber values, agricultural GHG flows, forestry GHG flows, and recreation). An interim objective (maxSVg) is also considered that is equivalent to maxMV, but includes values from GHG flows (i.e. excludes recreation).The present analysis omits non-monetised biodiversity constraints, but these will be included in future work.

5.7.3 Optimisation method

Market and non-market annuity values are summed during optimisation. For recreation-independent considerations, multiple approaches can be used to select the optimal location as these are independent across cells. For recreation, the interdependency between cells must be considered: planting a forest in one location means that an adjacent forest will garner less recreational utility for the population within its vicinity. The nonlinear function, approximated by a series of linear functions, this is mapped onto a linear program which is fed to a solver. All optimisations are constrained by the total number of hectares being planted. This is done on an annual time-step for the 50 year assessment period to produce optimal planting locations.

5.7.4 Planting locations and net benefits

Candidate optimal planting locations are identified under the three optimisation rules. Changes in value are calculated as a difference away from the BAU baseline, i.e. values presented are net of the underlying impact of changing climate.

Due to the effects of discounting, the long time between planting new trees and felling, and monies received from agricultural subsidies, the value of displaced agriculture (cost) always exceeds the value of net present timber production. The market, therefore, seeks to minimise the loss from agriculture by planting in the lowest value agricultural land available. Hence, if planting decisions are left to the market alone (maxMV), new forestry is planted in the extreme uplands (and away from urban populations; first panel, Figure 20).

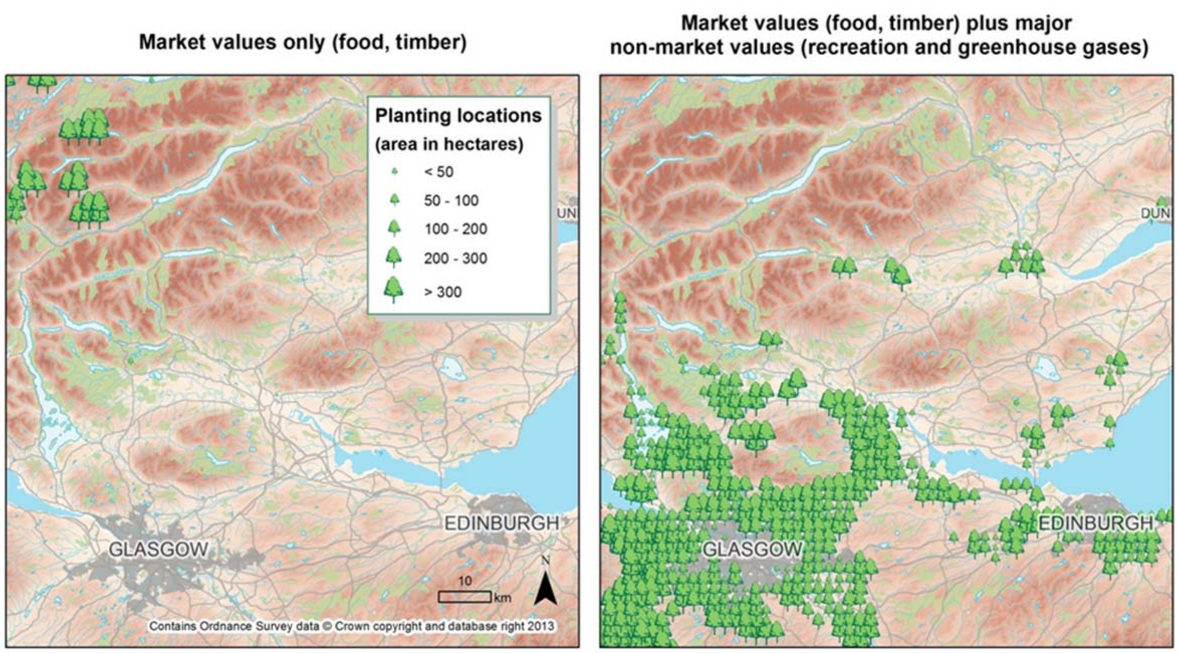

Figure 20. Candidate planting locations for Pedunculate Oak in Scotland's Central Belt under two approaches to decision making: maxMV (left panel) and maxSV (right panel) (upper carbon pricing regime)

Figure 20 shows the substantial shift in planting locations which occurs if GHG implications and recreation values are taken into account (maxSV). Almost all forests shift substantially from their previous locations. Several factors drive this move. On organic-rich soils, such as peat, soil carbon is liberated when the ground is disturbed and new trees are planted; away from peat soils, trees act as a net sink for carbon. Thus, stored soil carbon, and the potential for reducing GHG emissions by displacing livestock are important factors. However, the major driver here is the inclusion of recreation. This encourages new woodlands to be planted in accessible areas as near as possible to large populations (e.g. Edinburgh, Glasgow and Aberdeen). The Central Belt is particularly well-served by this strategy (second panel, Figure 20).

Although the afforestation policy can be implemented according to different rules (i.e. maxMV; maxSVg; maxSV), all market and non-market values from component systems are aggregated as societal consequences of this land use change. The bottom row of Table 18 shows that if non-market externalities are excluded from the assessment of optimal planting locations (i.e. maxMV) then the social value generated by the resulting forestry generates a net cost to society (negative). Conversely, if GHG flows and recreation influence planting decisions (i.e. maxSV) then substantial gains (£55 million per year) would be experienced by society. The disparity of costs and benefits is even more striking if an upper bound estimate for carbon pricing is considered.

The sensitivity of optimisation decisions to carbon pricing is shown by a comparison of Table 18 (lower carbon pricing regime) and Table 19 (upper carbon pricing regime). Investigation of these uncertainties is the focus of on-going work. Tables 18 and 19, while reporting differing magnitudes of total social value, demonstrate persuasively that including the natural environment in evaluating decisions can greatly improve value for money. Considering displaced agriculture alone results in planting in locations that provide a lower social value to society, while a more comprehensive assessment of the wider consequences of land use change can identify planting locations that generate major gains across society. Table 20 provides a regional breakdown.

Table 18. Conservative estimate (lower carbon pricing regime) of annuity values from planting Pedunculate Oak: Difference from BAU for Scotland (in 2013 £'s)

		Policy implementation method		
		maxMV	**maxSVg**	**maxSV**
Market Values	Agricultural Profits	-£1,716,269	-£1,773,355	-£67,896,012
	Timber Profits	-£26,737,108	-£26,730,855	-£25,577,719
	Total Market Value	-£28,453,377	-£28,504,209	-£93,473,732
Non-Market Values	Agricultural Carbon	£354,040	£342,193	£3,628,761
	Forest Carbon	£1,808,732	£1,971,001	£4,188,838
	Recreation	£13,422	£10,918	£122,249,012
	SPS (transfer to farmers)	£18,525,462	£18,520,786	£18,853,720
	Total Non-Market Value	£20,701,656	£20,844,898	£148,920,330
Social Value	**Total Social Value**	-£7,751,721	-£7,659,311	£55,446,599

Table 19. Upper estimate (upper carbon pricing regime) of annuity values from planting Pedunculate Oak: Difference from BAU for Scotland (in 2013 £'s)

		Policy implementation method		
		maxMV	**maxSVg**	**maxSV**
Market Values	Agricultural Profits	-£1,716,269	-£102,446,798	-£101,385,301
	Timber Profits	-£26,737,108	-£25,765,214	-£26,592,406
	Total Market Value	-£28,453,377	-£128,212,011	-£127,977,708
Non-Market Values	Agricultural Carbon	£6,424,594	£122,654,196	£109,948,757
	Forest Carbon	£29,415,388	£63,191,896	£69,393,880
	Recreation	£13,422	£20,463,034	£105,333,636
	SPS (transfer to farmers)	£18,525,462	£18,527,053	£19,287,431
	Total Non-Market Value	£54,378,867	£224,836,179	£303,963,704
Social Value	**Total Social Value**	£25,925,490	£96,624,168	£175,985,997

Table 20. Regional breakdown of annuity values from planting Pedunculate Oak: Difference from BAU (in 2013 £000's) (upper carbon pricing regime)

		maxMV			maxSVg			maxSV		
Region	**Region groupings**	**Market**	**Non-market**	**Social**	**Market**	**Non-market**	**Social**	**Market**	**Non-market**	**Social**
North East	NE Scotland	-£2,724	£5,688	£2,964	£0	£0	£0	-£779	£5,220	£4,441
North West	Eilean Siar	-£144	£278	£134	-£81	£71	-£10	£0	£0	£0
North West	Highland	-£21,717	£40,767	£19,049	-£24,245	£30,379	£6,134	-£16,183	£18,497	£2,314
North West	Orkney Islands	£0	£0	£0	£0	£0	£0	£0	£0	£0
North West	Shetland Islands	£0	£0	£0	£0	£0	£0	£0	£0	£0
South East	Fife	£0	£0	£0	£0	£24	£24	-£1,200	£5,024	£3,824
South East	Lothian	£0	£0	£0	£0	£26	£26	-£5,553	£25,251	£19,698
South East	Scottish Borders	-£15	£31	£16	-£456	£854	£399	-£179	£284	£105
South East	Tayside	-£2,504	£5,141	£2,637	-£625	£1,022	£397	-£1,320	£4,723	£3,403
South West	Argyll & Bute	-£1,044	£1,949	£905	-£31,654	£47,324	£15,670	-£20,720	£28,089	£7,369
South West	Ayrshire	-£28	£32	£5	-£20,410	£39,702	£19,292	-£24,026	£49,886	£25,860
South West	Clyde Valley	-£5	£3	-£2	-£18,038	£43,739	£25,701	-£33,947	£116,356	£82,409
South West	Dumfries & Galloway	£0	£0	£0	-£19,906	£37,161	£17,255	-£11,182	£20,286	£9,104
South West	East Central	-£273	£490	£217	-£12,798	£24,535	£11,737	-£12,889	£30,348	£17,459
	Scotland total	-£28,453	£54,379	**£25,925**	-£128,212	£224,836	**£96,624**	-£127,978	£303,964	**£175,986**

5.8 Extensions to current analyses

No appraisal of a complex system such as land will ever be absolutely complete. Similarly, a modelling exercise will always be, to some extent, an abstraction from reality. The criterion here is not to attain a perfect replication of land use and its determinants, but rather to deliver a robust analysis that reliably captures the major drivers of change and their associated trends. This research is undergoing continual refinement: from modifications to modules representing impacted systems to how this new approach of policy targeting (considering the natural environment) is presented to decision makers.

Modifications to component systems

Due to the modularity of the integrated approach, any component system can be removed, improved, added or replaced in a way that maintains consistency with any other system. This means that as more sophistication is added there is potential to optimise across a wider suite of social values and drivers of change. As with the nature of research, as more knowledge is amassed, modules are refined.

Dealing with uncertainty

Further research will consider a more robust optimisation methodology under conditions of uncertainty. The methodology will attempt to optimise when there are uncertainty bounds on the nominal annuity values; as an analogy consider stock portfolio selection where the aim is to seek an optimal value whilst limiting the downside risk as much as possible. Uncertainty in carbon price is an initial consideration.

Non-monetised constraints

An important extension of this research is to incorporate non-monetary constraints on policy and planting options. These could include, for instance, a requirement that any planting which reduces bird species diversity in an area be rejected (see for example, Bateman et al., 2013). In addition, constraints on water quality could also be considered.

Visualising policy options

The University of East Anglia houses a purpose-built Virtual Reality facility which offers seated VR viewing for up to 20 people, and comprises a 125° curved screen with three projectors. Active stereo presentations can be viewed using custom glasses, and support for interactive polling of participants is provided via wireless handsets. A control desk and PC allow a pilot to navigate around the displayed environments and control other displays such as presentations. This system is being tested to facilitate discussions with policy makers, and for public engagement.

5.9 Conclusions

The chief objective of this research is to contribute towards improved environmental-economic decision making to enhance the benefits derived from ecosystem services in the face of changing climate, land use and policy. The Integrated Model is introduced as an important tool for combining natural scientific underpinnings with environmental valuation techniques to identify benefit maximising policy options. We applied TIM to a policy-relevant case study devised in partnership with the UK, Welsh and Scottish Governments. The research reveals a number of important points for developing environmental policy:

- Continuing advances in climate, ecological and economic modelling mean that it is now possible to begin moving away from standard scenario analysis and towards genuine optimisation analyses of environmental policy. This is desirable for several reasons. First, making decisions on the basis of a small number of scenarios is severely limited by the fact that the best possible scenario may not be included in the set of considered options. Moreover, even if the best possible scenario is considered and pursued, there is no clear ‘road map’ for implementation.
- Because many of the benefits derived from the environment are generated and allocated outside of market systems, careful attention is needed to ensure their values are incorporated into decision making processes. The research here shows that failing to do so can lead to inefficient outcomes.
- Despite important developments in environmental valuation, not all streams of environmental benefits can be valued reliably. Biodiversity values in particular are notoriously difficult to assess consistently. However, economic analysis can still offer insight, for example by identifying the opportunity costs of biodiversity conservation policies.

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6 Developments of natural capital accounting

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6.1 Introduction

6.1.1 The concept of natural and ecosystem capital

“Natural Capital” (NC) is a term proposed by the British economist E. F. Schumacher in 1973, as a metaphor to shed light on the role of nature in supporting the economy and human welfare. The concept builds on the idea of manufactured capital as one of the factors of production (together with land and labour), which was introduced by Adam Smith and David Ricardo in the eighteenth century.

The term “capital” refers to a stock of materials or information, which can generate a flow of goods and services that improve human wellbeing. Ekins (1992) defines four kinds of capital, i.e., manufactured, human, social[14] and natural capital (see also Ekins, 2008), where the latter is constituted by the stock of natural assets that provide society with renewable and non-renewable resources (e.g., timber, water, fossil fuels, minerals) and a flow of ecosystem services. A five capitals model, developed by Forum for the Future during the 1990s and popularised by Porritt (2006) adds financial capital as a separate category. These capital stocks are in principle separately measurable, though in practice data are incomplete, and simplifying assumptions are necessary to derive simple measures at a national level for capital stocks that are in reality a combination of a vast array of complex elements. The methods presented in World Bank (2005, 2011) demonstrate the usefulness of the capitals model, breaking estimates of Total Wealth at the national scale into individual capital stocks, but the method does not distinguish between human and social capitals, and only accounts for parts of natural capital. The five capitals model has also been used successfully in simulation model of the integrated earth system, first in a non-spatial global model (GUMBO: Boumans et al., 2002) and subsequently in spatially-explicit modelling with MIMES (Boumans et al., 2015).

[14] Financial capital can be seen as part of social capital.

According to the analytical framework developed in the context of the EU 'Mapping and Assessment of Ecosystem and their Services' initiative (European Commission, 2013), Natural Capital includes stocks like sub-soil assets (geological resources) and abiotic flows like solar and wind energy. The Ecosystem Capital (EC) represents the biotic element of the Natural Capital and includes both ecosystems (which can be seen as stocks) and the flows of ecosystem services they provide to society (see Figure 21). This report will focus on the biotic components of Natural Capital, i.e., the Ecosystem Capital and the related ecosystem services.

However, it should be noted that the distinction between biotic and abiotic elements is not so clear-cut, as an ecosystem is "a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit" (Convention on Biological Diversity, 1992, art.2). For example, water is an abiotic element in itself, but ecosystems play a key role in its cycle, and also water is essential for nutrition and plays a key role in all ecosystems (Haines-Young and Potsschin, 2013). As another example, fossil fuels (an abiotic resource) were derived from the biological degradation of organic matter.

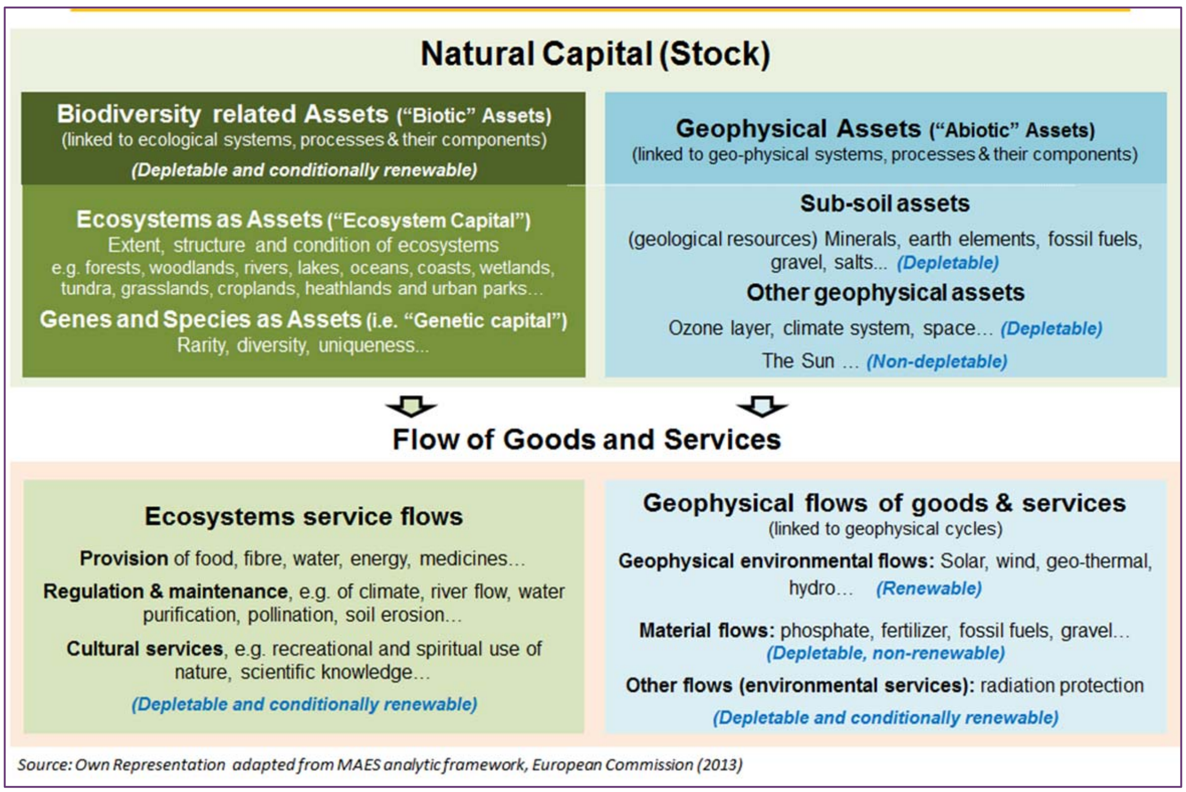

Figure 21. The components of Natural Capital and associated flow of goods and services

All four types of capital are needed to support human welfare. However, Natural Capital is arguably the most important one, as it is incorporated in all other forms of capital, and underpins them. Also, an important share of Natural Capital is non-substitutable with manufactured or other kinds of capital, and the manufactured, human and social capital would not be built without natural capital (Costanza et al., 1997). For example, minerals, metals and energy are needed to build the components of manufactured capital; human and social capitals are heavily dependent on the physical health of individuals, who in turn are dependent upon ecosystem services to maintain good health, including food, freshwater, timber and fibre and a wide range of regulating ecosystem services (e.g., water purification, nutrient cycling, protection from floods and other extreme events). In other words, the economy is embedded in the environment, and in order to be sustainable it needs to stay within its limits, both in terms of available resources and the capacity of the environment to absorb and process wastes.

The concept of Natural Capital is anthropocentric in nature, as it focuses on those aspects of nature that benefit humans, and makes no attempt to reflect the so-called ‘intrinsic value’ of nature or benefits to other species. However, in certain contexts it can play an important political role, as it can help to shed light on the benefits that nature provides to human society; and consequently on the need for nature protection not only for moral reasons but also as a way to enhance human wellbeing and economy. As such, it can contribute to influence policy-making towards an improved environmental protection, besides acting as an environmental education tool for awareness building. The benefits of anthropocentric values are that they can be known in principle, measured and integrated into decision making.

The natural capital concept also has risks – both practical and theoretical. Practical problems are that it will not be generally possible to known all the anthropocentric values of biodiversity and this might lead to bias (i.e., towards those that are easier to measure). The more theoretical problem of principle that could lead to problems of practice is that focusing only on benefits to society may lead to overlook the non-anthropocentric benefits. Both problems could be seen as encouraging the commoditisation of nature (McCauley, 2006; Kosoy and Corbera, 2010; Mace, 2014) and they may lead to prioritise the protection of areas and environmental resources that are more directly used by humans over others with greater biological diversity. For this reason, the Natural Capital concept needs to be seen in conjunction with wider biodiversity objectives and accounting needs to be used as a complementary tool to wider biodiversity indicators. Furthermore, it is important to understand to what extent accounts do (or could) take into account different types of natural capital, changes in the quantity and state of the natural assets, and the flow of associated ecosystem services, so as to understand the meaning of the accounts and how to interpret the outputs. This is a moving target as guidance and methods develop, as new data becomes available, and as initiatives at national (and subnational), EU and global scale improve our practices, tools, understanding and results.

6.1.2 Relevant initiatives for natural capital accounting at the European and global level

In recent years, there has been a growing interest in Natural Capital Accounting, which is reflected by recent international, European and national initiatives and legislation.

At the international level, the Strategic Plan for Biological Diversity 2011-2020 includes the commitment to integrate biodiversity into national accounting (Aichi Target 2), and commitments to accounting are also included in various National Biodiversity Strategy and Action Plans (NBSAPs). Also, a communiqué was issued at the 2012 Rio+20 Conference, supported by the EU and 57 countries to encourage the development of Natural Capital Accounting. In order to contribute to this process, the World Bank launched the Wealth Accounting and Valuation of Ecosystem Services (WAVES) Partnership, which aims to pilot methodological developments and experimentations with environmental accounts across the world[15], building on The System of Environmental-Economic Accounting (SEEA) developed by the UN Committee of Experts on Environmental-Economic Accounting (UNCEEA)[16], which provides detailed methodological guidance on how to prepare environmental-economic accounts (see next section on SEEA for details).

At the EU level, the first formal EU rules on environmental-economic accounting were established with Regulation 691/2011, which introduced the obligation for Member States to develop at least three kinds of accounts by 2013[17]: air emission accounts[18] (in physical terms), accounts on environmental taxes[19] (in monetary terms) and material flow accounts[20] (in biophysical terms). The Regulation establishes that more modules can be added in the future[21] to respond to key policy

[15] WAVES is funded by the European Commission, Denmark, France, Germany, Japan, the Netherlands, Norway, Switzerland, and the United Kingdom and it is being overseen by a steering committee. At the moment, the core WAVES countries - Botswana, Colombia, Costa Rica, Guatemala, Indonesia, Madagascar, the Philippines and Rwanda- are developing natural capital accounting.

[16] The UNCEEA is a body consisting of countries and international agencies under the auspices of the UN Statistical Commission.

[17] However, Spain, France, Cyprus, Malta, Austria, Poland were granted partial or total derogations and were allowed to present the accounts up to two years after the 2013 deadline.

[18] At least 14 different gases emitted by 64 industry groups and by households.

[19] Including at least four tax types – on energy, transport (other than fuel), pollution, and resources – all broken down into 64 industry groups, households and non-residents who pay these taxes.

[20] Material flow accounts for 50 material types showing domestic extraction, imports and exports. Then, Domestic Material Consumption = domestic extraction + imports – exports, for each type of material and in total.

[21] This is possible every three years, and the next window of opportunity is December 2016. The potential candidates for the next batch of modules are 1) Environmentally Related Transfers (subsidies); Resource Use and Management Expenditure Accounts (RUMEA); Water flow accounts; Forest Accounts, through the development of Integrated Environmental and Economic Accounting for Forests (European Commission, 2013).

needs; following this, an amendment[22] in 2014 added modules for environmental protection expenditures accounts, environmental goods and services sector accounts, and physical energy flow accounts.

The commitment to the development of physical and monetary environmental-economic accounts is also included in the 7th EU Environment Action Programme. In addition, the EU Biodiversity Strategy to 2020 requires Member States to map and assess the state of ecosystems and their services by 2014, and to assess their economic value and promote the integration of these values into accounting by 2020. In order to meet these commitments, the initiative ‘Mapping and Assessment of Ecosystems and their Services’ (MAES), was established by the European Commission, with support of Member States, the EU Joint Research Centre and the European Environment Agency (EEA). It aims to contribute to the mapping and assessment of ecosystems and ecosystem services, in biophysical, and in a later stage possibly also monetary terms, by providing a coherent analytical framework to the EU and Member States, and includes a module on Natural Capital Accounting.

Finally, the EEA is currently developing experimental Ecosystem Capital Accounts (ECA), based on the available data at the European level. The ECA process does not aim to generate new data, but to integrate the available ones at the European level. In order to do so, all utilised data sets are transposed into a 1km^2 grid across the entire area covered. The first experimental ECA will include land, organic carbon and water accounts.

Natural capital and environmental/ecosystem accounting initiatives are also being taken forward in some Member States. The UK in particular has developed work under the Natural Capital Committee, an independent advisory body set up to advise the Government on the sustainable use of natural capital. Their ‘State of Natural Capital’ reports[23] have presented evidence of significant economic and wellbeing benefits from better valuation and management of natural capital, highlighted where unsustainable use of assets place benefits at risk, proposed a long-term restoration framework, and recommended that the Government work closely with the private sector and NGO to develop a comprehensive strategy to protect and improve natural capital. The Committee has also worked with the major landowners (National Trust, Lafarge Tarmac, The Crown Estate and United Utilities) to advance corporate natural capital accounting and produce guidelines. The UK Office of National Statistics, meanwhile, has developed various satellite accounts[24] including environmental accounts, sustainable development indicators, and “initial and partial” estimates of the monetary value of natural capital.

[22] Regulation (EU) No 538/2014 of the European Parliament and of the Council of 16 April 2014 amending Regulation (EU) No 691/2011 on European environmental economic accounts
[23] https://www.naturalcapitalcommittee.org/state-of-natural-capital-reports.html
[24] http://www.ons.gov.uk/ons/taxonomy/index.html?nscl=Environmental+Satellite+Accounts

6.2 Natural and ecosystem capital and ES

6.2.1 The system of environmental-economic accounting (SEEA)

The System of Environmental-Economic Accounting (SEEA) provides detailed methodological guidance on how to prepare environmental-economic accounts. The first version was published by the United Nations Statistics Commission (UNSC) in 1993, and it was recently subjected to a wide revision process, led by the UN Committee of Experts on Environmental-Economic Accounting (UNCEEA), a body consisting of countries and international agencies under the auspices of the UN Statistical Commission. The revised version includes three volumes, as summarised in Table 21.

SEEA-Central Framework (SEEA-CF) - Volume 1- includes the biotic and abiotic stock and flows that cross the boundaries between the environment and human economy. It also covers typologies of environmental-economic accounts that are not part of Natural Capital Accounting, but can have a positive or negative impact on the Natural Capital, i.e., the environmental activity accounts, which include accounts for environmental protection expenditures, the environmental goods and services sector, environmental taxes and environmental subsidies. SEEA-CF provides standards for accounting that, when expressed in monetary terms, can be integrated into the System of National Accounts (SNA)[25] (the international standard for national economic accounts).

SEEA-Experimental Ecosystem Accounting (SEEA-EEA) - Volume 2 - covers accounts of ecosystems and ecosystem services. This kind of account is still at an experimental level, and for this reason, SEEA-EEA does not provide an internationally agreed standard for Ecosystem Accounting, but only a discussion on the methodological options and challenges, and general guidance on how to structure and develop accounts. The accounts included in the SEEA-CF and SEEA-EEA are to a certain extent complementary, as accounts included in the former provide useful information to describe the state of ecosystems (e.g., i.e., water accounts, timber accounts, land accounts) and the latter can offer insight on the state of ecosystems that provide the natural resources recorded in the SEEA-CF accounts.

Volume 3, Applications and Extensions of SEEA, shows some applications of SEEA data for their use in policy making and research, such as the use of environmental indicators and the analysis of environmental taxes and subsidies. It also includes an overview of the methodologies that can be used with SEEA data, and in particular the Environmentally Extended Input-Output Tables, a discussion on the spatial disaggregation of SEEA data and an overview on possible extensions of the SEEA to cover specific sectors and topics.

[25] SNA accounts are the main source of information for internationally comparable economic aggregates and indicators such as Gross Domestic Production (GDP), economic growth rate and government deficit.

Table 21. The SEEA guidance manuals

Publication	Year of publication	Scope	Standard	Possible integration into the SNA	Contents
Volume 1 Central Framework (SEEA-CF)	2012	Stock of natural resources, flows of natural resources towards the economy, their contribution to the economy and the impacts of economic activities on them.	Yes	Yes	1) Accounts of **flows** in physical terms for energy, water, material flows, air emissions, waste water and solid wastes. 2) Accounts of **assets** (in physical and monetary terms) for mineral and energy resources, land, soil resources, timber resources, aquatic resources, other biological resources and water resources. 3) **Environmental activity accounts** and related flows for environmental protection expenditures, the environmental goods and services sector, environmental taxes and environmental subsidies, in monetary terms. 4) **Combined physical and monetary accounts,** which provide the framework for the derivation of indicators such as resource efficiency and productivity.
Volume 2 Experimental Ecosystem Accounting (SEEA-EEA)	2013	The condition of ecosystems and the flows of ecosystem services.	No	No	1) Accounting for **ecosystem services** in **physical terms.** 2) Accounting for **ecosystem assets** in **physical terms** (**carbon** and **biodiversity accounts** illustrated more in detail). 3) **Main challenges** and **methodological options** for the **monetary valuation** of ecosystems and ecosystem services.
Volume 3 Applications and Extensions of SEEA	2014	Guide to the use of SEEA-based data in decision making, policy review and formulation, analysis and research. It includes the most common applications of the SEEA and possible extensions.	No	No	1) **Applications** of SEEA data, including the use of environmental indicators; the analysis of resource use and environmental intensity; the analysis of production, employment and expenditures relating to environmental activities; analysis of environmental taxes and environmental subsidies and similar transfers; analysis of environmental assets, net wealth, income and depletion of resources. 2) **Analytical techniques**: Environmentally Extended Input-Output tables (EE-IOT) and techniques for the analysis of input-output data (multiplier analysis; attribution of environmental pressures to final demand; decomposition analysis; computable general equilibrium analysis). 3) **Extensions of the SEEA**, including spatial disaggregation of SEEA data, extensions of SEEA to the household sector and to present environmental-economic accounts by theme (applied to the tourist sector as an example).

Source: own elaboration, based on the SEEA guidance manuals

Interestingly, whereas the MAES initiative and the European Environment Agency use the term “Ecosystem Capital Accounts” to define accounts covering both ecosystems and ecosystem services, in the context of SEEA, the wording “Ecosystem Accounts” is adopted, in order to underline that SEEA-EEA covers not only assets, but also flows. This chapter will adopt this convention.

Figure 22 provides a general overview of the different kinds of environmental-economic accounts and the role they can play in collecting and systematising the interactions between nature, society and the economy. The asset accounts included in the SEEA-CF measure the stock of Natural Capital (e.g., fossil fuels, minerals, timber, land) - generally in biophysical terms, but they can also be complemented by monetary information, if appropriate and where methodologies and data allow. The flow accounts included in SEEA-CF cover the flows of natural resources from the environment to the economy (i.e., inputs) as well as from the economy to nature (i.e. waste, water pollution and air pollution). SEEA-EEA accounts include both assets (ecosystem accounts) and flows (ecosystem services).

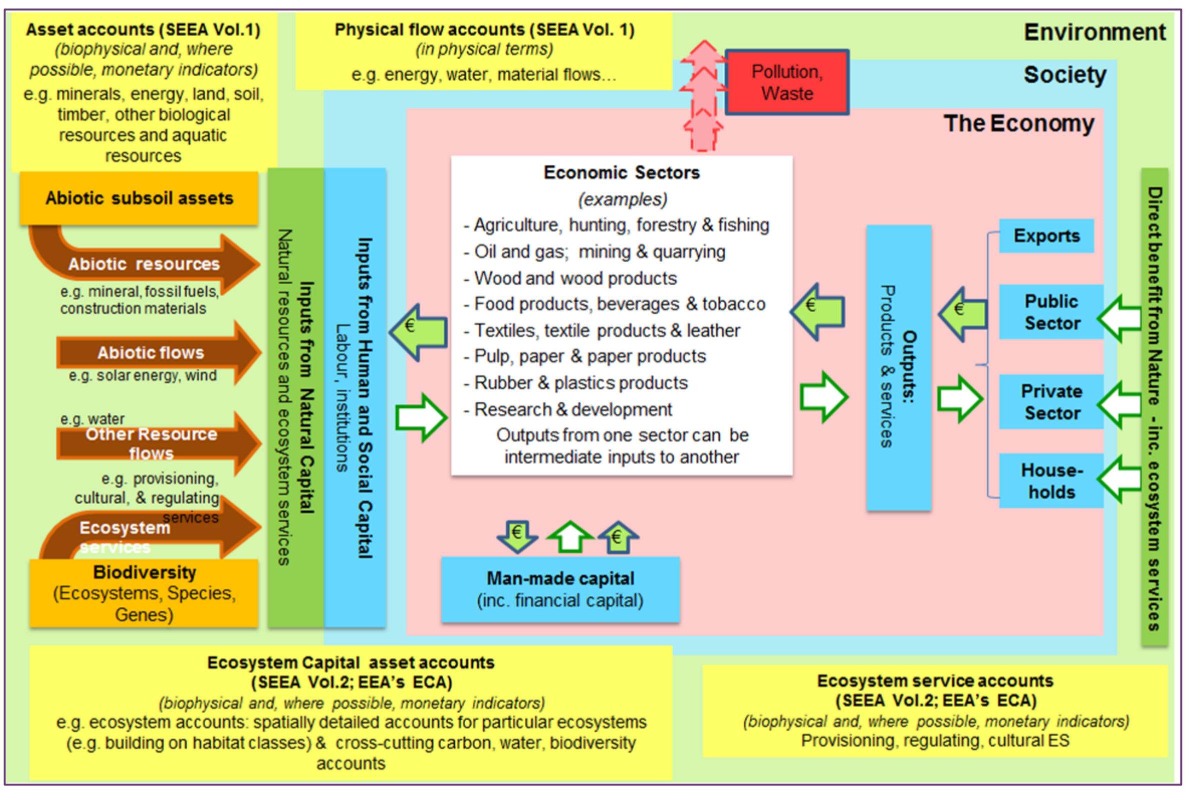

Figure 22. Environmental-Economic Accounts and Natural and Ecosystem Capital (source: adapted by ten Brink from Russi and ten Brink, 2013)

In principle, therefore, accounting should be able to integrate a wide set of natural capital types as well as flow of ecosystem services. In practice, data availability, limitations or lack of agreement on methods (i.e., still multiple experimental approaches being tested), and lack of actual development of accounts for some issues, means that there is only partial integration of natural capital and ecosystem services in accounts, with only a subset of issues represented in monetary terms. This underlines again the need to see the results of accounts in perspective of what they integrate and how. The section below looks at actual practice.

6.2.2 Examples of ecosystem accounting

Ecosystem Accounting is still at an early, experimental, stage and only a few examples have been developed at the national level. Brouwer *et al*. (2013) prepared a review of EU MS ecosystem service national assessments and found that for out of the 18 reviewed countries:

- Nine countries' - Austria, Belgium, Estonia, France, Germany, Hungary, Norway, Poland and Slovakia - national assessments are at the beginning stage (with no information on the valuation methods to be used);
- Three countries - Italy, Romania, Sweden - do not have on-going national assessments;
- One country - the Czech Republic - completed a study focussing only on ecosystem services provided by grassland (and using both methodologies based on costs and stated preference valuation);
- One country – Spain - completed a national Millennium Ecosystem Assessment, only in physical terms and without using monetary valuation so far;
- Only one country – Ireland - completed a study on benefits and costs of biodiversity, and used existing national value estimates and value transfer from UK;
- Only two countries - Lithuania, Netherlands - have on-going national assessments, the former using market valuation methods based on opportunity costs and possibly non-market valuation methods, and the latter aiming to use a wide range of methods, including market prices, cost-based pricing, contingent valuation, value transfer, travel costs and hedonic pricing methods.
- Only one country – the UK - has a fully-fledged National Ecosystem Assessment (UK NEA 2011), which covers all terrestrial and marine habitats, many provisioning, regulating and cultural services and a variety of monetary valuation methods (market prices, damage cost avoided, production function, stated preference, hedonic pricing, value transfer and replacement cost). The UK NEA involved a large number of researchers and entailed significant costs. The more recent UK NEA- Follow On exercise (2014)[26] incorporated further ecosystem services and developed the first UK-specific integrated model for valuing changes in multiple ecosystem services simultaneously.

[26] For more information, see http://uknea.unep-wcmc.org/

There has been considerable progress since 2012 in Europe and globally. In 2013, the UK Office of National Statistics have published experimental accounts and methodologies of UK land use, woodland area, timber resources, and woodland ecosystem assets and services.[27] France has regular forest accounts and is developing ecosystem accounts (EFESE), Portugal has been developing Marine accounts and Germany similarly developing national accounts that build in the concept of landscape ecosystem capacity (e.g., for soil). The European Environment Agency has been finalising their first generation Experimental Ecosystem Capital Accounts, and the EU's MAES process is finalising the EU reference document on Natural Capital Accounting (Petersen and Gocheva, 2015).

Globally, the World Bank's WAVES initiative is supporting natural capital accounting in five countries (Botswana, Columbia, Costa Rica, Madagascar and the Philippines). In addition, the CBD had developed guidance on Ecosystem Natural Capital Accounts (Weber 2014a), accounts have been public developed for Mauritius (Weber 2014b) and Madagascar, and a range of new initiatives are underway to support the development of accounts (e.g., TEEB initiative, supported by Norway), with plans to support NCA in Bhutan, Chile, Indonesia, Mauritius, Mexico, South Africa, and Vietnam.[28]

The above list will be expanded upon and key examples of interesting practice will be presented in detail in OPERAs Deliverable 3.4 in November 2015.

6.2.3 Summary: Status of integration of NC/ES in actual accounting practice

While Figure 22 provides a comprehensive overview of the different components of natural capital and ecosystem services, there exist constraints as regards the implementation of the concept. Some components of natural capital can be captured relatively well, as data is generally available and as the accounting units are accessible to observation, even though the methods of measurement undergo constant improvement. Among these are for example water quantity, carbon stocks in vegetation and soils, fish resources, or the extent of ecosystems. For other components of natural capital stocks, such a stock-taking appears possible in principle, but is constrained by data availability and an incomplete understanding of the natural biophysical and ecological processes underpinning the maintenance of natural capital and the production of ecosystem services. Once the data and natural scientific foundations are improved, such analyses will be possible, for example about the overall state of land ecosystems.

[27] http://www.ons.gov.uk/ons/rel/environmental/uk-environmental-accounts/2014/stb-stat-bulletin.html#tab-Experimental-natural-capital-accounts
[28] https://www.wavespartnership.org/en

Similar considerations apply to capturing the flow of ecosystem services. Some services such as the production of fish or local recreational values of landscape can be assessed with existing data and methods. In some cases like the services provided by wild pollinators, this is possible today, but an improved data basis is needed.

However, some aspects of natural capital are very difficult to capture, due to the characteristics of some of the stocks and flows. Marine ecosystems and water quality are examples of natural capital stocks that are difficult to capture in an accounting framework. In some cases, available methods do not allow reliable estimates at all, such as the complexity of ecosystems or the pool of genes.

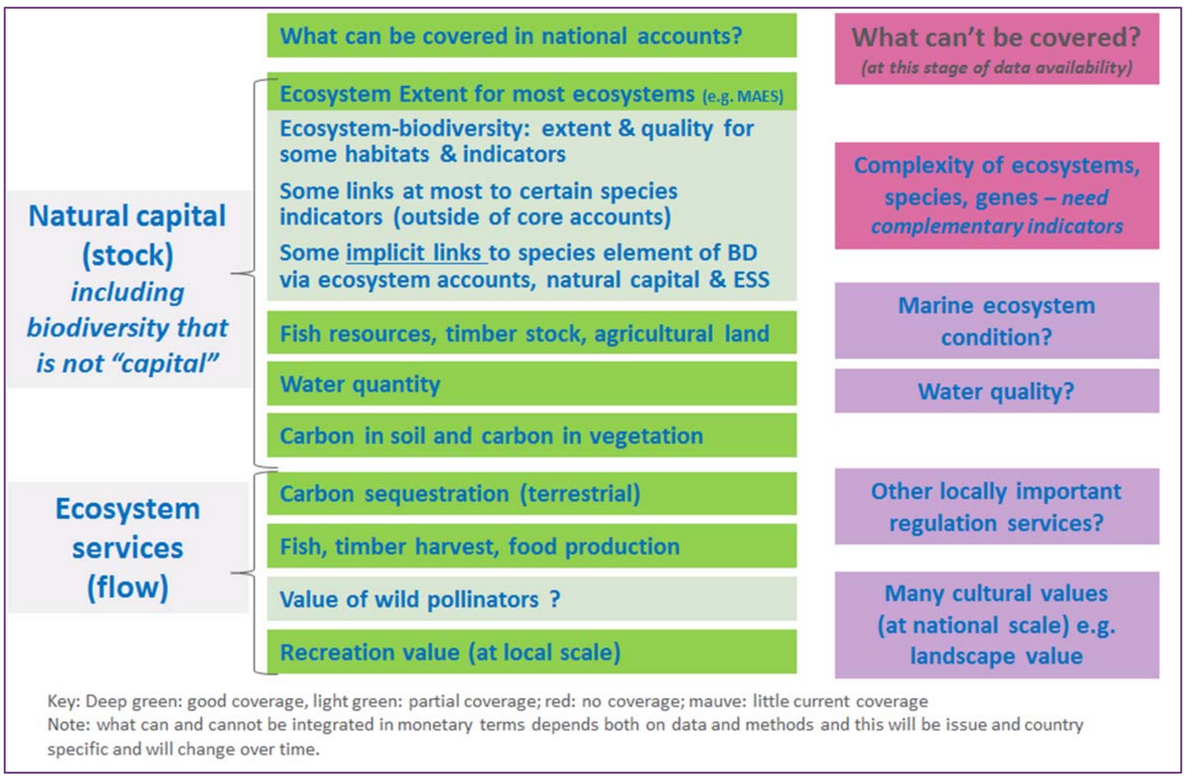

Figure 23. Feasibility of Economic Accounting for Natural Capital Stocks and Flows (source: own elaboration)

Brouwer *et al*. (2013) prepared a review of EU MS ecosystem service national assessments and found that most studies cover different kinds of provisioning, regulating, cultural and (in some cases) supporting ecosystem services, but only a small subset of them use monetary valuation methodologies to assess the ecosystem services. The study found that most provisioning services are or will be valued using market prices, and most regulating services using methodologies based on costs, where possible (see Chapter 2 for a fundamental critique of using cost-based monetary values). Monetary valuation of cultural ecosystem services, is much more complicated, because of methodological challenges, lack of data, lack of resources to conduct original valuation studies and also criticisms towards the use of monetary nonmarket valuation in some of the countries. However, the UK National Ecosystem Assessment Follow-on (2014) found that quantitative physical indicators of cultural ecosystem services can be developed using publicly available datasets.

6.3 Challenges for the development of ecosystem accounts and NC/ES integration in accounting

Ecosystem Accounts are still at an early stage of development, and, as explained above, only a few pilot experiments have been developed so far. This is partly due to a range of challenges that still need to be addressed.

One important challenge regards **data availability**. For many ecosystems and ecosystem services, significant data gaps represent an obstacle to the development of reliable accounts. In some cases, data may be available at a different scale than the one required for accounting, and therefore models and approximations need to be used. Also, it should be taken into account that data on some key ecosystems and ecosystem services may be very location specific, and for this reason they need to be translated into indicators relevant at the scale at which the accounts are developed, through an aggregation and extrapolation process. In some cases, accounts are compiled on the basis of a mixture of empirical data and outcomes of modelling exercises and in these cases data obtained through modelling should be compared, if feasible, with measurements taken in situ, in order to verify their robustness and reliability. It is important to remember that not all ecosystem services can be covered in Ecosystem Accounts, due to lack of data and methodological difficulties. For this reason, it is important to manage expectations, and find a balance between the demand for quick and easy indicators and for more detailed, time-intensive kind of accounts. It is also key to be transparent as to what accounts cover and clear on how to interpret the results. For example, accounts do not cover issues related to irreversible depletion or erosion of natural resources, ecosystems or ecosystem services in relation to ecological limits and thresholds (and nonlinearity), and in order to address these issues they would need to be combined with other analytical tools and data (Harris and Khan, 2013).

Another challenge to be addressed is the development of a coherent and agreed-upon **conceptual framework, methodology and definitions**. SEEA-EEA represented an important step in this sense, but since Ecosystem Accounting is still at an early stage, Volume 2 does not provide standards. For some of the most controversial topics, as for example monetary valuation, SEEA-EEA only offers an overview of the available methodologies and alternative definitions. The need for the development of a common vision on concepts and definitions is even more needed since many different typologies of experts are needed to develop and discuss accounts, including statisticians, economists, ecologists and hydrologists.

The **monetary valuation** of ecosystem services faces multiple methodological challenges due to the fact that many ecosystem services are not transacted in the market and for this reason do not have market prices. For this reason, economists have proposed three categories of methodologies to be used for monetary valuation of ecosystem services (see White et al., 2011, chapter 4 in ten Brink (ed.), 2011; Pascual et al., 2010, chapter 5 in Kumar P. (ed.), 2010; see also Brouwer *et al.*, 2013, table 4 and SEEA Central Framework, Chapter 5):

1) Methodologies based on **costs**, which use market prices to indirectly estimate the monetary value of ecosystem services. Examples include methodologies based on the **avoided costs**, such as the economic damage from floods by managing floodplains in a sustainable way; methodologies based on the **replacement cost**, such as the cost of mechanical purification of water, which is needed to replace natural water purification provided by healthy ecosystems; and methodologies based on the **restoration costs**, which are the cost of restoring a degraded ecosystem (but see chapter 2 for a fundamental critique of using cost-based value estimates).
2) Methodologies based on **revealed preferences** estimate values based on the preferences of individuals, shown by their behaviour. Examples are the **Travel Cost Method** and the **Hedonic Pricing Method**. See Chapter 2 for a detailed discussion on these methods.
3) Methodologies based on **stated preferences** such as **Contingent Valuation** and **Choice Experiments** use the preferences that are directly stated by people through surveys. They investigate people's willingness to pay (WTP) for improved environmental conditions or their willingness to accept (WTA) compensation for a reduction in environmental quality (see also Chapter 4).

Also, since monetary valuation studies are time and resource intensive, in many cases monetary values already calculated elsewhere for similar ecosystems are used. This procedure is called "**value (or benefit) transfer**" (see Chapter 2) and needs to be carried out very cautiously because the provision of ecosystem services are often location-specific (see White et al., 2011, in ten Brink (ed.), 2011; Pascual et al., 2010, in Kumar P. (ed.), 2010; Brouwer et al. 2013, section 6.2.4.3, SEEA Vol2,section 5.6.3; and Kettunen and ten Brink (ed.), 2013).

There is an on-going debate as to whether to use methodologies based on **costs**, which employ market prices to indirectly estimate the monetary value of ecosystem services (e.g., estimates of the avoided economic damages from floods ensured by sustainable floodplain management or estimates of avoided water pre-treatment costs for municipal drinking water provision) or methodologies based on **individual preferences**, based on for example on surveys that investigate people's willingness to pay for improved environmental conditions (Brouwer *et al.*, 2013). In general, the methods based on revealed and stated preferences are based on the measurement of changes in individual welfare, whereas accounts are based on the exchange value.

For example, Weber (2011) states that for environmental accounting, monetary valuation should be carried out on the basis of **restoration costs**[29] because he considers monetary valuation methodologies based on stated or revealed preferences as incompatible with environmental accounting, because they are based on **subjective evaluations, which make up-scaling and aggregation disputable**. On the contrary, others maintain that because revealed preference techniques make use of real world, actually observed behaviour, they avoid charges of subjectivity that are sometimes valid criticisms against stated preference studies (Bateman et al., 2011; Bateman et al., 2014). Moreover, advances in benefit transfer methods (see Chapter 1 and Bateman et al., 2011) can offer some response to disputes over up-scaling and aggregation. Finally, the methods applied throughout the UK NEA, for instance, maintained that restoration and replacement costs should not be used as proxies for the economic value of ecosystem services. Ecosystem service values reflect the **change in the stakeholders' wellbeing due to a marginal change in the provision of ecosystem services**. This is not dependent on what is arguably the exogenous cost of restoration. Moreover, restoration costs reflect technological ability rather than the value of an environmental asset: if a technology was developed that reduced restoration costs by 50%, it does not necessarily follow that the value of the asset has also been cut by half.

Using valuation methods aimed at identifying the impact on welfare of changes in ecosystem services (i.e. methods based on stated or revealed preferences) implies to include the consumer surplus, i.e., the difference between the price consumers are willing to pay for a good or service and the market price. Cost benefit analyses include the consumer surplus in the monetary valuation of environmental goods and services, but this is not coherent with the with the SNA approach, which is based on market prices. This point will need further discussion among experts.

SEEA-EEA allows both categories of valuation methodologies to be used (i.e., the ones based on preferences and including the consumer surplus and the ones based on costs), but warns that if methodologies based on preferences are used, some adjustments need to be done (e.g., using shadow prices) (see SEEA-EEA, Chapter 5 for more details on this discussion).

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[29] Restoration costs are the costs to restore an ecosystem to its original state before degradation. Complications arise in that restoration rarely gets the ecosystem back to the original state – i.e. ex post actual costs may not be a fully adequate measure. Similarly, assessing degradation costs for not restored areas poses significant methodological challenges as it require assumptions on how the restored ecosystem should be. An added complication is the choice of the re-introduced species, as if degradation has gone beyond the ecological tipping point restoration can be very expensive (or even de facto infinite in price in the case of extinct species). Finally, the cost of restoration is only a proxy of value and depending on the context can be an over-estimate or under-estimate. Each of these elements poses important challenges.

A 'third way' option is provided by the concept of 'simulated exchange values' (Caparrós Gass & Campos Palacín 2009; Oviedo, 2010) which estimates the value of ecosystem services in terms of potential revenue if a market were to exist. This arguably represents a more consistent basis for including their value in national accounts alongside monetary transactions, because consumer surplus is excluded. The method aims to measure the income that would occur in a hypothetical market where ecosystem services were bought and sold. It involves estimating a demand and a supply curve for the ecosystem service in question and then making further assumptions on the price that would be charged by a profit-maximising resource manager under alternative market scenarios. The method then takes the hypothetical revenue associated with this transaction (excluding the associated consumer surplus) as a measure of value of the flow of ecosystem services. It should be noted, however, that this drives a wedge between the quantity of ecosystem service associated with the valuation (at the intersection of supply and demand curves) and the quantity actually observed. For example, with a paid market in recreation, one would expect lower numbers of visits than when access is free. This has the potential to add confusion between the monetary and physical accounts.

Other related issues are whether and how to **aggregate results obtained with different methodologies** and how to **scale up results** obtained through valuations at the local level. In general, if different methodologies are used for monetary valuation (such as in the UK NEA), the outcome values of different ecosystem services may not be fully comparable or compatible (as they may measure different things in different units) or additive, and care will be needed to avoid double counting, interpreting of meaning and aggregation. To be additive requires, inter alia, the value of a given hectare of land and its interaction on the value of other hectares of land need to be factored into account (new facilities for recreation in one park may increase its recreation value but also reduce the recreation value in other parks (Kettunen and ten Brink 2013) (see chapter 5). This may pose a problem if monetary valuation is to be used for accounting purposes, as different units are used in accounting (market exchange values) and welfare economics (Brouwer *et al.*, 2013).

Another problem related with monetary valuation based on stated or revealed preferences is the fact that people may not be aware of the ecosystem services they benefit from (typically in the case of regulating ecosystem services). For this reason, stated preference (SP) techniques should arguably only be used for end-services (though values for regulating services can be derived from SP for end-services, e.g., benefits of reduced flood risk can shed light on regulating services of flood control).

Also, the high costs related to data collection and processing usually represent an obstacle for monetary valuation of natural capital. Furthermore, though experts agree on the principle of discounting and the formula to be used, they do not agree on how to derive the parameters (Arrow et al, 2013), and therefore do not agree on the discount rate to be used for the valuation of natural resources[30].

Finally, **gaps in the scientific evidence base** regarding the key biophysical and ecological processes that replenish natural capital and generate ecosystem services remain a key challenge for environmental accounting.

In summary, many challenges as regards integrating monetary aspects of natural capital in accounting remain and national experimentation is crucial to be able to highlight potential promising ways forward. This issues Economic valuation methods – responding to the needs of national accounting is addressed in more depth in Chapter 2.

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7 Further research

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7.1 Introduction

This deliverable is aimed at providing an overview of the state-of-the-art in monetary and social valuation of ecosystem services. We addressed the available monetary and social valuation methods (Chapters 2 and 3), and presented developments in monetary valuation (Chapter 4), in integrated modelling of ecosystem service values (Chapter 5) and in natural capital accounting (Chapter 6). This final chapter aims to provide a concise overview of the plans for further research on these topics within OPERAs, and on the application of some of the methods and tools within the OPERAs exemplars.

7.2 Economic valuation

With respect to economic valuation within OPERAs and the OPERAs exemplar, various research activities will take place. These activities have been described extensively in Milestone 3.3 (Koetse and Brouwer, 2014) and in Milestone 3.9 (Koetse et al., 2015). We briefly discuss these activities below.

Incorporating spatial heterogeneity in meta-analysis value transfer functions

While value transfer may provide a quick and cheap alternative to original valuation research, some conditions must be met if it is to provide reliable results (see, e.g., Bad'ura et al., 2015; Bateman et al., 2011). Above all, the local circumstances and conditions in the new decision-making context need to be closely related to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explanatory variables into the transfer. We follow up this suggestion by including in the meta-analyses databases the spatial characteristics of study areas. These characteristics are obtained from external resources, generally using geographical information systems (GIS). A first goal is to collect existing and build new meta-analyses databases, and incorporate spatially explicit information in these databases in order to arrive at spatially specific value transfer functions. Examples of spatial variables we aim to include in these meta-analyses are income of population in study area, population density of study area, and overall scarcity and supply of good or service of study. The meta-analysis databases we aim to expand and develop are a meta-analysis database of water values and a meta-analysis database of forest values.

For further details we refer to OPERAs Milestone 3.3, which contains the research design for incorporating spatial heterogeneity in meta-analysis databases and value functions. The goal is to apply the spatially explicit value functions in exemplars aimed at ES valuation at a larger scale. For details we refer to Milestone 3.9, which discusses the application of the value functions in the Scottish, European and Global exemplars.

Ecosystem service valuation in local exemplars

In exemplar aimed at ecosystem service valuation at a more local scale it is less sensible to apply value transfer from meta-analysis functions, if only because the local context is so situation-specific that value estimates obtained from meta-analysis value functions are too unreliable. We therefore aim to do primary valuation studies in these exemplars.

For example, within the Swiss exemplar a choice experiment has been performed. The choice experiment was aimed at eliciting how local residents value and trade future changes in regionally relevant ecosystem services with a focus on cultural services. In the survey, participants had to choose between the current landscape and two alternative future landscapes, each landscape being described by a set of ecosystem services. The results are used to show how future changes in ecosystem services compromise or improve the utility of the landscape for the participants.

Within the Montado exemplar the goal is also to do a choice experiment, which is aimed at discovering use and non-use values of the Montado, and to assess whether Montado landowners are willing to change their production practices, and if so, whether and how much they would need to be compensated for realising this change. In this exemplar different samples of respondents (visitors, general population, Montado landowners) are used to address the different questions.

Finally, the study in the Inner Forth exemplar contains several elements that are of interest for both economic and socio-cultural valuation. This study aims to explore how coastal realignment in the Inner Forth would affect the delivery and value of services using ecological, sociocultural and economic valuation techniques. From a valuation perspective the most relevant research questions is how learning and group deliberation shape preferences and values for coastal ES changes? To answer this question we perform a valuation study among citizens the area, eliciting monetary values and preferences for future coastal changes through a choice experiment. For our main purpose we distinguish between a choice experiment in a workshop context and through a more or less standard one-to-one survey. The impact of learning and social deliberation on preferences and values for coastal ES change will be measured in a workshop where participants complete choice tasks three times: prior to learning, after learning, and after learning and deliberation. The one-to-one survey will be used as a control treatment. For further details we refer to OPERAs Milestone MS 3.9 (Koetse et al., 2015), which discusses the research design for ES valuation in the Inner Forth, Swiss Alps and Montado exemplars.

Additional case study on revealing hypothetical bias

One of the more problematic issues in economic valuation through stated preference research is that of hypothetical bias. Hypothetical bias refers to the potential problem that by asking people what that *would be willing to pay*, e.g., through contingent valuation or choice experiments, people can basically state or choose anything without actually having to take account of their household budget. We aim to measure this effect through a study on valuation of biodiversity and landscape development in the Netherlands. One of the aims of the study is to address this hypothetical bias in economic valuation. In this respect, the most relevant research question is what is the impact of including real payments on the willingness to compensate? To answer this question we perform a contingent valuation study on the willingness to donate to a fund from which land is bought for a farmer to either (1) switch to an extensive and biodiversity enhancing production method, or (2) use part of his land for introducing landscape elements. The research design includes split samples for different treatments. These treatments are aimed at assessing the extent of hypothetical bias in stated preference research, and of identifying the potential mitigating effects of including a cheap talk script. For more detail we refer to OPERAs Milestone 3.9 (Koetse et al., 2015).

7.3 Socio-cultural valuation

The strongest argument for incorporating socio-cultural valuation into the overall valuation of ecosystem services is that it allows for a wider perspective on values than economic approaches alone. Economic methods, including non-market valuation, are most useful where the good in question includes a substantial utility contribution to personal or group welfare. Other considerations such as emotional motivations or values attached to equity or the welfare of others are only represented to the extent that they are captured within the individual’s utility function. Very often they may be, but the monetary element that necessarily enters into economic valuation inevitably influences the decision making framework. This may divert a respondent’s attention to monetary trade-offs or, conversely, cause them to refuse to accept such a trade-off.

The socio-cultural valuation approach has been described in Milestone 3.4 (Bullock et al., 2014) and Chapter 3 of this report. It recognises that motivations such as equity or aesthetics can have an equal role in determining preferences to utility based motivations and that there can be a gradient between values that are commensurable with one another and values that are incommensurable. Group valuation methods are commonly used whereby individuals are brought together to *deliberate* on a subject using workshops or citizen juries. These provide a forum whereby social learning can be nurtured by a greater understanding of other people's point of view often with equitable results. In addition, deliberation provides participants with more time to learn about complex issues such as ecosystem services than the postal or one-to-one surveys favoured by standard economic valuation methods.

For an overview of applications of socio-cultural methods in selected exemplars we refer to Milestone 3.10 (Bullock et al., 2015). Below we discuss some of the issues that will be addressed in these studies. The question of complexity is an issue in both the Firth of Forth and Fingal (Dublin) exemplars with the former looking to address the issue of sea level rise and the latter looking to examine attitudes towards the regulating ecosystem services provided by less popular amenity destinations such as estuaries. One hurdle we are attempting to overcome in Fingal is people's natural tendency to bundle different types of benefits, including values that relate to ecosystem services with others that do not. An Ecosystem Approach requires people to distinguish between these different types of goods, but because people's interaction with the environment is often conducted through manmade infrastructure such as recreational facilities, this is not an easy task. This has been an issue in the public workshops in Fingal where stakeholders have expressed understandable concerns about aspects of coastal management that do not necessarily relate to ecosystem services. In addition, respondents may bundle together biotic and abiotic goods, in which the latter may have no or just a small contribution from ecosystem services. In the Fingal exemplar we have therefore been using a networked system diagram to help participants explore the relationship between complementary goods, goods with a high ecosystem services connection, and others for which the connection is lower or non-existent. Such diagrams have been used successfully by others, e.g., Kenter (2014), although we have adapted the network to bring in both tangible and less tangible cultural service benefits.

Mapping has also been a feature of the Fingal and Pentland Hills (Edinburgh) exemplars. Both are using deliberation to identify areas of high socio-cultural value. Inevitably, high values are attached to the more well-known or accessible locations, but deliberation can also be used to demonstrate how these values, along with the knowledge or awareness that underpins them, varies with the type of participant. In Fingal this process has provided us with additional information about the coast and threats to ecosystem services of which we might not have been aware. In the Pentlands, mapping has been combined with a process to identify individual motivation types.

An asset of the alternative of economic valuation is its ability to capture quantitative values using the metric of money, a metric that can be used to provoke trade-offs between alternative environmental goods. Single values can be aggregated easily in the process. By comparison, deliberative methods can be very valuable in providing for learning and additional information, but are also difficult to extrapolate to a larger sample. This report has described how hybrid methods, such as deliberative monetary valuation, can be used to chart changes in stated willingness-to-pay as the deliberation progresses and participants gain a greater understanding of the issue and of other people's values. However, the method still has the limitation of requiring a monetary vehicle to denote values and so OPERAs is experimenting with trade-off scenarios that do not involve a direct monetary attribute.

Alternative quantitative methods can be brought into the deliberation through the use of scoring and weighting approaches rather than monetary estimates. In principle, it can be difficult to ensure that participants adopt a common scale for scores and weights. In this respect, each of the exemplars is applying trade-offs to one degree of another to provide more consistency of relative value. In the Firth of Forth deliberation will be combined with economic choice experiments. In the Pentland Hills an algorithm has been used to force survey respondents to weigh up the relative value that they attach to different park and landscape attributes. In Fingal, areas of weak incommensurability (between the extremes of strong and no commensurability) are being mapped to identify areas in which deliberation can be used to achieve comparable outcomes to economic valuation by encouraging participants to debate and trade-off values and objectives. In these cases, there is the potential for participants to consider trade-offs between the attributes of a good or, indeed, between economic and other motivations.

In Fingal, our objective is to use deliberative exploration of socio-cultural values in line with the guidelines provided by TESSA (Partner UNEP-WCMC) as a means to inform future local decision making. We are exploring the use of participatory multi-criteria analysis (PMCA) as a means of tracking the progression of the deliberative process in a manner similar to that applied by Garmendia and Gamboa (2012), but at the level of strategic spatial planning. This is, however, an approach that requires firm ground rules and structure. Such formality can be at the expense of valuable insights into local knowledge and the qualitative exploration of fundamental values. Providing a replicable workable method that is both informed and informative and can be trusted by local decision makers is the challenge that we are aiming to overcome.

7.4 Integrated modelling

Recent years have seen the development of a series of integrated ecosystem service modelling, mapping and valuation tools. Bagstad et al. (2013) review several, including ARIES, Co$ting Nature, EcoServ, InVEST, LCUI, MIMES, and SolVES. These vary in terms of their technical sophistication, the spatial and temporal scales they consider and the degree of expertise needed to use the model. While the growing number of integrated models is a good sign for a field in its infancy, it can also be confusing for end users, particularly when models and decision support tools yield differing results. This highlights three key issues in the field of integrated ecosystem service modelling:

- **No individual model is best for all applications.** Differing results are driven by fundamental differences in each models' overall scope and underlying assumptions, as well as differences in the range of ecosystem services considered, spatial resolution, and the type and quality of data used. Those currently developing integrated models should pay particular attention to developing sensitivity to spatial heterogeneity in both the biophysical and human (i.e. demographic and socio-economic) components of the model. Further challenges include forecasting beyond the range of observed data, addressing aggregation effects and dealing with uncertainty.
- **Fitness for purpose – answering policy relevant questions.** Arguably, one of the primary uses of integrated ecosystem service modelling is to provide a sound scientific basis for improving environmental decision making. To this end, integrated models must be able to address real world policy questions, which often differ substantially from scientific research questions. Going forward, developers and users must engage early in the process to ensure models are fit for purpose. This entails facing several practical challenges, such as being able to adjust core assumptions and parameter values within the model (e.g., carbon prices, discount rates, etc.) to match government guidance and the needs of end-users. More broadly, substantial challenges remain in making models accessible to non-specialists. An important step forward would be to simplify the computational, informational and training requirements for using models, as well as reducing the time taken to run.
- **Integrating natural sciences into economic modelling.** Reporting the impacts of environmental change in monetary units simplifies and clarifies relevant trade-offs in a way that is accessible to decision makers from all backgrounds. However, environmental valuation may only ever be as robust as the natural science upon which it lies. While many existing integrated models include natural science and economic elements, very few can genuinely claim to integrate them robustly. Moreover, only The Integrated Model (Bateman et al., 2014) incorporates scientific and economic modules within a formal optimisation routine.

Despite substantial advances in integrated modelling, several specific challenges remain. With respect to data, there is a need to expand awareness of, access to and utilisation of existing data sets. Although the need for new data is inevitable, research in the short to medium term can be improved (and costs reduced) by making better use of existing information. Beyond data concerns, there are a range of more fundamental issues that model developers must address. First, models need to expand their scope to incorporate a wider range of ecosystem services (particularly those concerning air quality and human health) as well as international issues such as trade and global climate policy. This would entail deliberately designing new, theoretically consistent process-based natural science models regarding health and air quality, but also identifying the international implications of domestic policies (transboundary rivers, for instance, may raise particularly interesting questions and applications). Inevitably, this would require the incorporation of more sophisticated feedback loops to better reflect the myriad interdependencies that characterize environmental-economic interactions. Finally, integrated models should attempt to move beyond scenario analyses towards genuine optimisation. This would support the development of more comprehensive decision support tools, ultimately making research more readily accessible to the end-users and policy uses.

To highlight the potential of integrated modelling to incorporate and value changes in multiple ecosystem services, The Integrated Model (TIM) was used to assess the impact of introducing a new afforestation policy in Scotland. The afforestation policy used in the case study was developed in partnership with Scottish Parliament and consists of planting 5,000 ha of new forest each year from 2014 to 2063. The primary policy question was “where should Scotland plant its new forests?”. The research modelled expected changes in agricultural production, agricultural greenhouse gasses, timber production, forestry greenhouse gasses, and outdoor recreation, all of which were valued in economic terms. Water quality and biodiversity were also included, but not valued directly. The model was run for three different optimisation criteria:

- Planting trees to maximize market values;
- Planting trees to maximize market and greenhouse gas values;
- Planting trees to maximize market, greenhouse gas and recreation values.

The analysis and results are explained in further detail in Chapter 5 of this deliverable, however the overarching conclusion was that the optimal planting location for new woodlands changes substantially when multiple ecosystem services are taken into account. This highlights the need for integrated models that incorporate and value a wide range of services, as decisions made on the basis of partial analysis are very likely to result in poor value for money.

7.5 Developments with Natural Capital Accounting

With respect to accounting, Chapter 6 in this deliverable is an interim product. Ongoing work on natural capital accounting and valuation will be reported in D3.4 (Ten Brink et al., 2015, The use of (economic & social) values of NC/ES in national accounting, expected in November 2015). This deliverable is foreseen to include 4 chapters:

- Chapter 1: National accounting and the extent of integration of NC/ES (responsible: IEEP);
- Chapter 2: Integrating social values of EC/NC in accounting (responsible: University College Dublin)
- Chapter 3: Chapter 4: Integrating economic values of NC/ES into national accounting (responsible: UEA and Iodine);
- Chapter 4: Ecosystem Accounting, the Integration of economic and social values of EC/NC and implications for policies, decision making and instruments (responsible: IEEP).

Ongoing and next steps include:

Step 1: Literature review and expert interviews on the case examples used to illustrate case applications of accounts and also policy utility – this will support Chapter 1 in D3.4 in particular (cases), but also help with Chapter 4 (policy utility).

Step 2: Explore the role of economic tools to help integrate economic and social values of EC/NC into accounting – this will help with Chapters 2 and 3 in particular.

Step 3: Interview with key policy makers as regards current and future policy utility of accounts – this will help with Chapter 4 in particular.

Step 4: In parallel to the above: present at and attend expert workshops and conferences to help gain information on practice, identify practitioners and relevant policy makers, and disseminate OPERAs work.

Step 5: Write up the papers, drafts, expert review and finalisation: Nov 2015.

Step 6: Journal publication –Special section of the journal *Ecosystem Services*: for late 2106.

Within the wider Exemplars, there is ongoing discussion as to whether some accounting approaches could be used to illustrate the scale of natural capital in the Global exemplar, and the scope for exploring the policy utility of accounting in the European and Scottish exemplars.

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