EEA Report | No 4/2010

Scaling up ecosystem benefits

A contribution to The Economics of Ecosystems and Biodiversity (TEEB) study

ISSN 1725-9177

European Environment Agency

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Cover design: EEA
Cover photos: © iStockphoto
Layout: EEA

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Luxembourg: Office for Official Publications of the European Union, 2010

ISBN 978-92-9213-098-5
ISSN 1725-9177
DOI 10.2800/41295

Environmental production
This publication is printed according to high environmental standards.

Printed by Schultz Grafisk
— Environmental Management Certificate: ISO 14001
— IQNet – The International Certification Network DS/EN ISO 14001:2004

— Quality Certificate: ISO 9001: 2000
— EMAS Registration. Licence no. DK – 000235
— Ecolabelling with the Nordic Swan, licence no. 541 176

Paper
RePrint — 90 gsm.
CyclusOffset — 250 gsm.
Both paper qualities are recycled paper and have obtained the ecolabel Nordic Swan.

Printed in Denmark

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Acknowledgements

This EEA Technical report was drafted by Onno Kuik of the Institute for Environmental Studies (IVM) and Hans Vos of the European Environment Agency (EEA). Valuable comments on the final draft were received from Helen Dunn of the United Kingdom Department for Environment, Food and Rural Affairs (DEFRA).

This report was based on the results of two previous studies commissioned by EEA. The first report, 'Scaling up ecosystem services values: methodology, applicability and a case study — Report to the European Environment Agency under contract 3603/B2007/EEA.53106 (2008)', was prepared by Paulo Nunes, Anil Markandya and Andrea Ghermandi of Fondazione ENI Enrico Mattei (FEEM), and Luke Brander, Onno Kuik, Marije Schaafsma and Alfred Wagtendonk of IVM.

Results from this first study were discussed at an expert workshop at EEA headquarters on 28 April 2008. The discussions greatly helped to advance the present project and EEA is indebted to the participants for their support.

The second study, 'Scaling-up ecosystem services values: towards guidelines for a policy-relevant approach — Report to the European Environment Agency under contract 3603/B2008/EEA.53405 (2009)', was prepared by Onno Kuik, Luke Brander, Marije Schaafsma, Alfred Wagtendonk of IVM; and Ingo Braüer, Holger Gerdes of Ecologic; and Ståle Navrud of Universitetet for miljø- og biovitenskap (UMB).

The second report benefited greatly from comments from the Project Supervisory Board Members Helen Dunn (DEFRA), Anil Markandya (FEEM), Berta Martin-López of Universidad Autónoma Madrid (UAM), Paulo Nunes (FEEM), and by Aude Neuville (European Commission DG Environment).

Final editing and production were done respectively by Mike Asquith and Henriette Nilsson (EEA).

The EEA project manager was Hans Vos.

Summary

At the 2007 meeting of the environment ministers of the G8+5 in Potsdam, Germany, the European Commission launched The Economics of Ecosystems and Biodiversity (TEEB) study. Its aim is to assess the economic repercussions of global biodiversity loss.

TEEB has reinforced the need for the economic valuation of changes in ecosystems at large geographical scales. Assessing the costs and benefits of changes in ecosystems includes the valuation of scarce, non-market goods and services. That requires the use of specialised research methods that are commonly labour intensive because they frequently involve interviewing and detailed statistical analysis. Such techniques are often location-specific and become expensive and time-consuming when carried out across large geographical areas, including multiple ecosystem sites.

The costliness of economic valuation studies has led researchers to consider using data from existing primary research in novel ways. The research question that the present study addresses is if and how existing data on the economic value of non-market ecosystem services can be used through *value transfer*, taking into account the location, size, scarcity and other attributes of the individual ecosystem sites, the proximity of residential areas, and the purchasing power of (potential) users or other beneficiaries of the ecosystems.

Commonly value transfer takes primary data from one ecosystem site — the 'study site' — and applies them at another single and similar site — the 'policy site'. Scaling up builds on the methods and tools for value transfer by taking economic values from a particular study site (or sites) and extrapolating them to a larger geographical area.

The present report analyses options for scaling up existing estimates of ecosystem service values to larger geographical scales. It also presents a case study of wetlands at the European level and discusses the results and policy applications.

The case study looks into ways to improve large-scale assessments by applying scaling up. The study assesses the economic value of a historical change in wetlands in the Netherlands and the Baltic states using a meta-analytic value transfer function with coefficients for wetland size, wetland scarcity, per capita income and population density. The analysis concludes that the gains and losses in the study period (2000–2006, as determined by the availability of Corine Land Cover maps) more or less cancel each other out.

Based on this research, the report discusses the results of applying scaling up in a policy context, for example in TEEB. Successfully applying scaling up with the help of value transfer methods requires that the policy context be clearly and properly defined. No scaling-up exercise will ever be able to answer a question such as 'what is the value of all wetlands in Europe?' Scaling up may, however, help in answering a question like 'what is the benefit of halting wetland loss in Europe in comparison to a trend of continuing wetland loss over the next twenty years?'

The present report stresses the importance of natural scientific knowledge. If, in a specific area, there is a lack of scientific knowledge about important relationships between environmental pressures, ecosystem functioning, and the provision of ecosystem services, neither economic valuation nor scaling up will add anything to our understanding of these relationships.

Scaling up makes it possible to combine (several sets of) primary data and one or more value transfer methods to assess the economic value of changes in ecosystem services at a larger spatial scale. The magnitude of the change under study affects the direct applicability of values taken from primary research.

Primary valuation studies usually assess the values of ecosystem services under the assumption that all else would remain equal. A small change in

ecosystem service provision (e.g. the loss of a small area) will not affect the value of services from other ecosystem sites. Non-marginal changes in ecosystem service provision, however, will affect the value of services from the remaining stock of ecosystems. As the ecosystem service becomes scarcer, its value will tend to increase. Moreover, scaling-up exercises must take account of cross-substitution effects between ecosystem services and diminishing returns to scale.

Value transfer and scaling up can generate substantial errors. These may be limited by carefully addressing potential measurement and generalisation errors and publication biases but they can never be totally avoided. Maximum acceptable (transfer) errors may differ from case to case.

Cost-benefit analyses of particular policy options or damage assessments for use in court will generally require a high level of accuracy. By contrast, less detail is normally needed for broad impact assessments of proposed policies or regulations, or studies that aim to underline the need for policy action in general terms, to prioritise between different policies (cost of inaction studies) or to raise awareness.

In the end, when primary data are too limited for a scaling up exercise — based on criteria to be developed — any value transfer method may lead to unacceptable transfer errors. In such circumstances, value transfer is not a viable option and primary research is necessary for a reliable outcome.

1 Introduction

1.1 The need for ecosystem valuation

There is wide political agreement that biodiversity loss must be significantly reduced or halted. The European Union (EU) formulated the ambitious goal of halting biodiversity loss by 2010 in its Biodiversity Action Plan (EC, 2006). In 2010, the European Commission published options for an EU vision and target for biodiversity beyond 2010 (EC, 2010), stressing the importance of managing, maintaining and enhancing ecosystem functions that provide services for society at large.

To assist in developing biodiversity policies and strategies, decision-makers are demanding ever more information on the economic implications of losing nature and biodiversity. According to the Millennium Ecosystem Assessment (MEA, 2005), biodiversity loss is expected to threaten the potential welfare of future generations. More exact information on the benefits and costs of achieving global and regional goals through effective policies and conservation is, however, largely lacking.

The 'Stern Review' (Stern, 2007) has proven to be a key element in louder and more widely acknowledged calls for more effective climate change policy. It suggests that relatively modest investments today could prevent far more costly economic damage in the future. In response to calls for a comparable analysis of the costs of biodiversity loss and benefits of preventive actions, various initiatives have been launched, such as The Economics of Ecosystems and Biodiversity (TEEB, 2010).

TEEB published its first report in 2008 (TEEB, 2008). It analyses the possible loss of biodiversity by 2050 according to a 'business as usual' scenario and concludes that '11 % of the natural areas remaining in 2000 could be lost. Almost 40 % of the land currently under low-impact agriculture could be converted into intensive agricultural use, with further biodiversity losses. 60 % of coral reefs could be lost — even by 2030.' To help alter that trend, TEEB sets itself an ambitious task. Its ultimate aim 'is to provide policymakers with the tools they need to incorporate the true value of ecosystems services into their decisions.' Recognising that 'ecosystems economics is still a developing discipline', TEEB (2008) identifies a number of common messages for developing the economics of ecosystems and biodiversity, including, 'measure the costs and benefits of ecosystems services.' These few words summarise a task of enormous complexity, as the report itself illustrates.

Ecosystems goods and services provide the basis for life on earth in many different ways. These include direct support such as delivering food and shelter (crops, fish, meat, water, fuels, materials), protection and health (flood defence, pollution absorption, medicines) or pleasure (variety of species, landscapes). They also include indirect means of support, such as regulating nutrients, water and carbon, via complex and largely unknown dynamics of symbiosis between living organisms and their environment. Without ecosystem goods and services, life would be impossible. If a little disappears, life may continue undisturbed but further degradation will at some point start to disrupt society.

The value of a particular ecosystem service is not constant but varies depending on many conditions, including scarcity, quality, access, wealth of the users and availability of alternatives. Where ecosystems goods and services are abundant, they are less vulnerable; losing a few hectares of forest in, for example, Finland will do little damage in general. Where services are already scarce, however, what remains is of high value. The last wells in a desiccate area are thus more precious than diamonds. Given the localised character of ecosystem valuations, how is it possible to value the loss of biodiversity and ecosystem services on a larger, even global scale?

The TEEB interim report aims at such an aggregate assessment but stops short of a comprehensive economic valuation. It mentions the economic causes of biodiversity loss, including market and policy failures. It also describes important elements of an economic valuation framework, including the ethical choices affecting intra-generational and intergenerational equity, and risks and uncertainties.

The TEEB report points to the extent of knowledge needed to undertake monetary valuation. We need to know how biodiversity underpins ecosystems services, how changes in biodiversity would affect the quality and resilience of these services, how affected services would change in a quantitative sense, before we can put a price on the loss. This is quite a challenge at the local scale, let alone on the regional or global scale, and all the more so because many services are not traded and have no market price. Much of this information is not available and some never will be. Hence, assessing the economic damage due to lost biodiversity will always be an 'educated guess' at best. This is not, however, a reason to cease valuation attempts. Rather, efforts must focus on further improving the education behind our guess in order to support policymakers in their decision-making.

One attempt to expand our knowledge base, cited by TEEB, is 'The Cost of Policy Inaction (COPI): The case of not meeting the 2010 biodiversity target' (Braat *et al*., 2008), which was written for the European Commission. That report contains an economic assessment of the value of biodiversity loss in 2050 compared to 2000, according to a business-as-usual scenario. It arrives at monetary results but cites numerous caveats, making the results partial and tentative. One such caveat is the absence of ecosystems goods and services' values in a proper, quantitative dimension This is a severe barrier to reliable estimates because the large majority of these values have been derived from local case studies. They have only local significance because they depend on specific conditions, which vary from location to location. A global assessment would thus require a huge number of values for specific services in specific contexts ([1]). Very few values have wider relevance without adaptation, although examples do exist, such as the value of carbon sequestration because climate change is a global phenomenon.

In a second phase TEEB endeavours to provide further relevant information for target groups who need to prepare and implement biodiversity policies: policymakers at the national and local level, the business community and consumers.

There is a great need for either up-scalable values or methods that allow local data to be scaled up without adaptation. This report aims to contribute to meeting that need by analysing methods that adapt local values in such a way that they can be used on a larger geographical scale. The report also discusses proper combinations of such methods in concrete cases. It builds on the research results already available in the area of value transfer and it combines the use of location-specific parameters and spatial grid analysis.

1.2 Outline of the present report

This report analyses methods for scaling up existing estimates of ecosystem service values to larger geographical scales (e.g. the European scale), illustrates the methods with a case study, and discusses the results in a policy context. Chapter 2 describes the concepts of value transfer and scaling up, and the importance of spatial scale in valuing ecosystem services. Chapter 3 briefly introduces the concept of ecosystem services. Chapter 4 surveys the literature on value transfer methods as important building blocks for scaling-up exercises. Chapter 5 presents options for scaling up, discussing their strengths and weaknesses. Chapter 6 illustrates the options with a case study on the economic value of a historical change in wetland area in the Netherlands and the Baltic states. Based on this research, Chapter 7 discusses applying scaling up results in the policy context. Chapter 8 discusses the contribution the report's findings can make to the valuing of ecosystems services benefits as initiated with the TEEB programme.

([1]) The COPI study, Annex 1, states that putting a distinct value on the 19 different ecosystem services it identified in the context of 13 biomes in 14 geographical regions would produce a list of 27 000 separate values. Within its partial analysis of four biomes the study was able to detect only 30 values that it could usefully apply in its estimate.

2 Value transfer and scaling up

Political initiatives like TEEB have reinforced the need for the economic valuation of changes in ecosystems across large geographical scales. Assessing the costs and benefits of changes in ecosystems includes the valuation of scarce, non-market goods and services. That requires the use of specialised research methods that are commonly labour-intensive (often requiring household surveys, data processing and detailed statistical analysis) and hence expensive and time-consuming, especially if they are carried out across larger geographical areas, including multiple ecosystem sites.

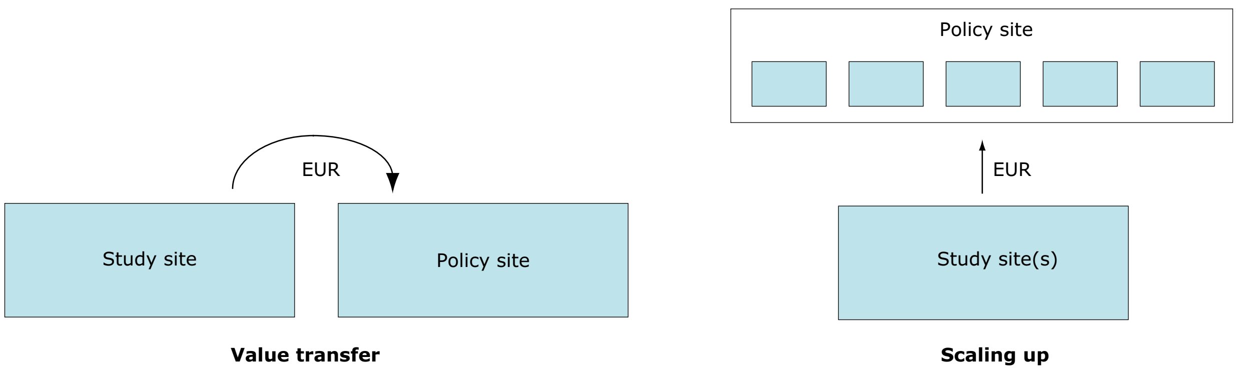
These challenges have led researchers to consider using data from existing primary research results (e.g. on the economic value of European wetlands) in novel ways. The research question for such studies is if and how existing data on the economic value of ecosystem services can be utilised for large-scale assessments through *value transfer*, taking into account the location, size, scarcity and other attributes of the individual ecosystem sites, the proximity of residential areas, and the purchasing power of (potential) users or other beneficiaries of the ecosystems.

2.1 Scaling up

The approach of using existing data on economic values of local ecosystem services for an assessment of these values at a larger geographical scale can be termed 'scaling up'. In a scaling-up exercise, economic values from a particular study site are transferred to another geographical setting, for instance to the regional, national or global scale. Local values are thus not applied in another local context, but are used to estimate the values of all ecosystems (or ecosystem services) of similar characteristics in a larger region.

Scaling up builds on the methods and tools that have been developed for value transfer, and can be seen as an extension of value transfer. Value transfer is usually applied on a case-by-case basis. The transfer of economic values of individual ecosystem services from a particular study site to another — but similar — site (the policy site) has become a common tool in ecosystem assessment. In the scaling-up exercise, economic values from a particular study site (or sites) are extrapolated to a larger geographical setting (Figure 2.1).

Figure 2.1 'Scaling up' and 'value transfer'



2.2 Spatial scale

Spatial scale is recognised as an important issue in valuing ecosystem services (Hein *et al*., 2006). The spatial scales at which ecosystem services are supplied and demanded contribute to the complexity of ecosystem valuation and management.

On the supply side, ecosystems themselves vary in spatial scale (e.g. small individual patches, large continuous areas, regional networks) and provide services at varying spatial scales. The services that ecosystems provide can be both on and off site. For example, a forest might provide recreational opportunities (on site), downstream flood prevention (local off site), and climate regulation (global off site).

On the demand side, beneficiaries of ecosystem services also vary in terms of their locational distribution. The spatial scale over which ecosystem services are provided and received is determined by the spatial scale over which an ecosystem function has effect and the spatial scale of (potential) beneficiaries. To conceptualise the relationship between the supply of and demand for ecosystem services, one might imagine two overlaid maps — one representing the spatial extent of an ecosystem and the (potential) services it provides, and the other representing the spatial location of the (potential) beneficiaries of these services. It is important to recognise that ecosystem services result from the interaction of ecosystem functions and human activities. An ecosystem does not provide a service if no one makes use of its potential to provide that service.

Ecosystem services often have different groups of beneficiaries (in terms of spatial location and socio-economic characteristics). For example, the provision of recreational opportunities by an ecosystem will generally only benefit people in the immediate vicinity, whereas the existence of a high level of biodiversity may be valued by people at a much larger spatial scale. Differences in the size and characteristics of groups of beneficiaries per ecosystem service need to be taken into account in aggregating values for each service. The management of ecosystems may be further complicated in cases where the interests of different groups of beneficiaries (possibly at different spatial scales) are in conflict. This may occur when ecosystem services are mutually exclusive (e.g. timber extraction and carbon sequestration).

The values that beneficiaries ascribe to ecosystem services may vary due to a number of different factors that can be spatially defined (distance, availability of substitute and complementary sites, income, culture and preferences). Use values are generally expected to decline with distance to an ecosystem — so called 'distance decay'. Non-use values may also decline with distance between the ecosystem and beneficiary, although this relationship may be less related to distance than to cultural or political boundaries ([2]). The availability of substitute (complementary) sites within the vicinity of a selected ecosystem is expected to reduce (increase) the value of ecosystem services from that ecosystem. Socio-economic characteristics of beneficiaries (e.g. income, culture, and preferences) are not spatial variables per se, but differences in these variables between (groups of) beneficiaries can often be usefully defined in a spatial manner (e.g. by administrative area, region or country).

Consideration of the spatial scale of the provision and beneficiaries of ecosystem services is important for calculating the total economic value of these services (i.e. the aggregation of values across relevant areas and populations). In addition, accounting for spatial scale may also be useful in formulating policies to manage ecosystem services, for example in identifying winners and losers, the need for compensation or incentives, and the design of policies such as payments for environmental services.

Regarding the estimation of ecosystem service values, several important issues should be considered related to spatial scale. In discussing these scale-related issues, the present study distinguishes between estimating values for an individual ecosystem site and for the entire stock of an ecosystem within a large geographic area. The latter case is referred to as 'scaling up' ecosystem values when a lack of data necessitates value transfer methods.

At the level of an individual ecosystem site, unit values for ecosystem services are likely to vary in accordance with the characteristics of the ecosystem site (area, integrity and type of

([2]) The difference between use and non-use values is explained in Chapter 3.

ecosystem), the beneficiaries (number, income, preferences) and context (availability of substitute and complementary sites and services). All of these variables have a spatial dimension that can be accounted for in estimating site-specific values. For example, in terms of ecosystem area, many ecosystem service values have been observed to exhibit diminishing returns to scale (i.e. adding an additional unit of area to a large ecosystem increases the total value of ecosystem services less than an additional unit of area to a smaller ecosystem). For some services (e.g. recreation opportunities in forests or flood defence by coastal marshlands) an ecosystem might have a minimum size below which it stops providing that service. It is therefore important to account for the size of the ecosystem being valued.

To scale up ecosystem values to estimate the total economic value of a change in the stock of ecosystems in a large geographic area, in addition to controlling for other spatial variables, it is necessary to account for the non-constancy of marginal values across the stock of an ecosystem. At the margin, a small change in ecosystem service provision (e.g. the loss of a small area) will not affect the value of services from other ecosystem sites. Non-marginal changes in ecosystem service provision will, however, affect the value of services from the remaining stock of ecosystems. As the ecosystem service becomes scarcer, its marginal and average values will tend to increase. This means that simply multiplying a constant per unit value by the total quantity of ecosystem service provision is likely to (substantially) underestimate the total value of a negative change. Appropriate adjustments to marginal values to account for large-scale changes in ecosystem service provision need to be made.

3 Ecosystem services

Why is an assessment of the values of ecosystem services necessary and why is there growing demand from environmental policymakers for valuation studies? Environmental resources are valuable to our society because they provide us with certain benefits, and a change in environmental quality accordingly results in a change in social welfare (Kahn, 2005).

Collectively, these environmental benefits are described as ecosystem services. The Millennium Ecosystem Assessment (MEA, 2005) grouped these ecosystem services into provisioning, regulating, cultural and supporting categories. Valuing the benefits that our society obtains from these services is of central importance for environmental policy formulation, as it puts the costs of achieving certain environmental objectives into perspective.

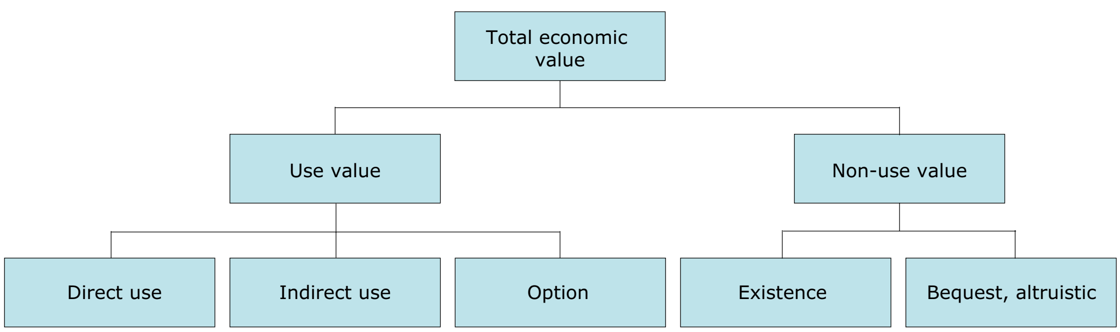
Terrestrial and aquatic ecosystems provide a large array of benefits to human society. Wetlands, for instance, provide freshwater for residential and agricultural use; retain nutrients and thus improve water quality; help to control floods; provide food for humans; preserve biodiversity; and provide numerous recreational opportunities. The sum of these environmental and resource benefits is described as 'total economic value' or TEV (Figure 3.1).

In determining total economic value, one distinguishes between direct use, indirect use, option, and non-use values. While direct use values arise from direct interaction with the natural resource (e.g. agricultural irrigation, fish harvesting, recreational swimming), indirect use values are associated with services (e.g. flood protection, removal of pollutants) but do not entail direct interaction. Ecosystem services may also be valued for their potential to be used in the future, i.e. they have an option value. In addition to these use values, there exist non-use values, which do not arise from direct or indirect usage (i.e. existence values, bequest values and altruistic values).

Society thus benefits from the actual or potential use of environmental goods and services, either in a direct or indirect way. In trying to attach economic values to these goods and services, one is faced with the challenge that most of the services that the environment provides are not captured in commercial markets, with the implication that it is difficult to quantify the value in monetary terms. While the value of marketed goods can be determined by means of existing prices, the valuation of non-marketed goods proves more difficult. However, since it is important to know the total economic value of an ecosystem, economists have developed special techniques for measuring the value of non-marketed goods.

Economists emphasise the importance of distinguishing between functions and services for valuation purposes. Ecosystem *functions* can be

Figure 3.1 'Total economic value' and its components



defined as the capacity of ecosystems to supply goods and services, while ecosystem *services* is the flow of goods and services that is actually provided. While both functions and services could be valued in principle, Turner *et al*. (2003) prefer the valuation of services because some functions are impossible or very difficult to value ([3]).

Ansink *et al*. (2008) argue that one-to-one mapping between functions and services is not always possible, as one function can add to the supply of several services and one service can depend on several functions. They therefore emphasise consistency in ecosystem valuation: in order to avoid neglecting values and to avoid double counting, either functions or services should be valued, but not both.

In practical terms, services are easier to value than functions. Fisher *et al*. (2009) argue that the appropriate classification of ecosystem services should be based on the characteristics of the ecosystem and on the decision context for which it is used.

As an example, Table 3.1 presents a classification of wetland ecosystem services (Brander *et al*., 2006). It distinguishes between ecological/physical ecosystem functions, their associated economic goods and services (ecosystem services), and the type of value derived.

Despite the increasing interest of policymakers and researchers in 'ecosystem services' or 'ecological services', no 'agreed upon, meaningful and consistent' definition of ecosystem services exists (Fisher *et al*., 2009). Different authors and studies have used different definitions and classification schemes.

Table 3.1 Ecological functions, ecosystem services and types of value

Ecological function	Ecosystem service	Value type
Flood and flow control	Flood protection	Indirect use
Storm buffering	Storm protection	Indirect use
Sediment retention	Storm protection	Indirect use
Groundwater recharge/discharge	Water supply	Indirect use
Water quality maintenance/nutrient retention	Improved water quality Waste disposal	Indirect use Direct use
Habitat and nursery for plant and animal species	Commercial fishing and hunting Recreational fishing and hunting Harvesting of natural materials Energy resources	Direct use Direct use Direct use Direct use
Biological diversity	Potential future use Appreciation of species existence	Option Non-use
Micro-climate stabilization	Climate stabilization	Indirect use
Carbon sequestration	Climate change mitigation	Indirect use
Natural environment	Amenity and aesthetic Recreational activities Value associated with leaving natural environment for future generations	Direct use Direct use Non-use

Source: Adapted from Barbier, 1991; Barbier *et al.*, 1997; Brouwer *et al.*, 1999; and Woodward and Wui, 2001.

([3]) This is especially true for those functions that ensure the healthy functioning of the system, notably the 'glue' that holds the ecosystem together (its self-organising capacity). Turner *et al.* (2003) argue that this 'glue' or infrastructure might possess a form of 'insurance value' that is 'both highly significant and yet formidably difficult to value' (Turner *et al.*, 2003, p. 498).

4 Value transfer

4.1 Value transfer methods

Value transfer is the procedure of estimating the value of an ecosystem (or goods and services from an ecosystem) by borrowing an existing valuation estimate for a similar ecosystem. The ecosystem of current policy interest is often called the 'policy site' and the ecosystem from which the value estimate is borrowed is called the 'study site'. This procedure is often termed 'benefit transfer' but since the values being transferred may also be estimates of costs or damages, the term 'value transfer' is arguably more appropriate.

The use of value transfer to provide information for decision-making has a number of advantages over conducting primary research to estimate ecosystem values. From a practical point of view it is generally less expensive and time consuming than conducting primary research. Value transfer can also be applied on a scale that would be unfeasible for primary research in terms of valuing large numbers of sites across multiple countries. Value transfer also has the methodological attraction of providing consistency in the estimation of values across policy sites.

The methodological design of primary valuation studies (e.g. valuation method, elicitation format, payment vehicle) have been shown to have a significant influence on the values estimated. In the absence of standardised applied methodologies across primary studies, value transfer offers a means of estimating values that do not reflect methodological differences.

Value transfer methods can be divided into four categories:

- unit value transfer
- adjusted unit value transfer
- value function transfer
- meta-analytic value function transfer.

Unit value transfer involves estimating the value of an environmental good or service at a policy site by multiplying a mean unit value estimated at a study site by the quantity of that good or service at the policy site. Unit values can be expressed as values per household, values per activity day (recreation), or as values per unit of area. Total values are calculated by multiplying these unit values by the number of households that benefit from the good or service, by the number of activity days (e.g. fishing days), or by the total area.

Adjusted unit transfer involves making simple adjustments to the transferred unit values to reflect differences in site characteristics. The most common adjustments are for differences in income between study and policy sites and for differences in price levels over time or between sites.

Value function transfer methods use demand or value functions estimated through valuation methods (travel cost, hedonic pricing, contingent valuation, or choice modelling, as set out in Table 4.1) for a study site together with information on parameter values for the policy site to transfer values. Demand or value functions for environmental services commonly include parameters such as income, age, gender and education. Parameter values of a policy site are plugged into the value function to calculate a transferred value that reflects the characteristics of the policy site.

Meta-analytic value function transfer uses a value function, estimated from multiple study results together with information on parameter values for the policy site, to estimate policy site values. The value function therefore does not come from a single study but from a collection of studies. This allows the value function to include greater variation in both site characteristics (e.g. socio-economic and physical attributes) and study characteristics (e.g. valuation method) that cannot be generated from a single primary valuation study.

Table 4.1 Economic valuation methods

Travel cost	The travel cost (TC) method is used to estimate economic use values of ecosystems or sites that are used for recreation. The travel cost method assumes that the time and travel expenses that people incur to visit a site can be viewed as the 'price' of access to the site. As this 'price' will differ for different people (e.g. because of the length of the journey), it is possible to construct a demand schedule relating the number of visits (demand) to the travel costs (price). Peoples' willingness to pay to visit the site can then be deduced from the demand schedule.
Hedonic pricing	The hedonic pricing (HP) method can be used to estimate economic values for ecosystem services that directly affect market prices. A common application is to infer the value of local ecosystem services from variations in house prices. The method requires the estimation of a statistical function that relates property value to property characteristics, including environmental characteristics (a beautiful vista or the proximity to a recreational forest).
Contingent valuation	The contingent valuation (CV) method is a survey-based method that can be used for valuing ecosystem services. In a CV survey, respondents are asked how much they are willing to pay for the provision of an ecosystem service in a hypothetical market. Essential elements of the survey are: description of the service that is to be valued, description of the payment vehicle (the way that the respondent is hypothetically supposed to pay for the service) and description of the hypothetical market (who will provide and who will pay). The method is called 'contingent' valuation because people are asked to state their willingness to pay, contingent on a specific hypothetical scenario and description of the environmental service.
Choice experiments	The choice experiments (CE) method is also a survey-based method. An ecosystem site, for example a forest, is described by a number of characteristics or attributes. Attributes could include things like availability of a visitor centre, length of walking tracks, number of rare species of plants and animals, and entrance fee. By varying attribute levels, the CE analyst can create several hypothetical alternatives. In a sequence of choice tasks, respondents are asked to choose their most preferred alternatives. As each alternative has different attribute levels, by choosing respondents implicitly make trade-offs between the levels of the attributes in the different alternatives. Thus, they indirectly reveal their relative preferences for different attributes. If one of the attributes is a price (like, for example, an entrance fee), relative preferences or utility can be expressed in money terms. Like contingent valuation, it is a hypothetical method — it asks respondents to make choices based on a hypothetical scenario. But it differs from contingent valuation because respondents are not asked to state their preferences in money terms. Instead, values are inferred from the choices or tradeoffs that the respondents make.

Source: Adapted from the website 'Ecosystem Valuation', www.ecosystemvaluation.org.

The unit value transfer method is relatively simple and transparent but it has the obvious problem that individuals at the study site may not value the good in question in the same way as the individuals at the policy site (Kristófersson and Navrud, 2007). This may be due to differences in the characteristics of the population (e.g. income, age, gender), or differences in the overall supply of the good: at the study site the good may be scarce, while it may be abundant at the policy site (Kirchhoff *et al*., 1997).

Other transfer methods try to adjust for these differences, to varying extents. The adjusted unit transfer method makes simple adjustments to some characteristics (e.g. income), and the value function and meta-analytic value function transfer method go further by estimating a function that is meant to 'explain' values at the policy site in terms of observable characteristics of the ecosystem (service) and the population at the policy site. From a theoretical perspective, the 'function' approaches are to be preferred to the 'unit value' approaches. The questions remains, however, of how they perform in practice. As the proverb says, 'the proof of the pudding is in the eating.'

4.2 Tests of value transfer methods

Fortunately, an emerging body of studies have tested value transfer methods in the area of ecosystem services. A number of such studies will be discussed with the aim not of giving an exhaustive overview of the subject but rather to give some insight into the practice of value transfer, its potential and limitations, and its accuracy ([4]).

Brouwer and Spaninks (1999) test value transfer between two areas of peat meadow land in the Netherlands, one area in the province of Friesland (in the north of the country) and the other in the province of South Holland (in the west of the country). The services provided by these sites concern the preservation of wildlife habitat (rare meadow birds and flowery ditch-side vegetation) through agricultural wildlife management. Primary

([4]) For a more in-depth discussion of the validity and accuracy of value transfer, see Navrud and Ready, 2007.

valuation studies had been carried out in both areas, using comparable (contingent valuation) methods. Both studies had estimated how much households neighbouring the sites were willing to pay per year to compensate farmers for the wildlife conservation measures. The mean willingness to pay in Friesland was lower than in South Holland (55 and 74 florins respectively per household per year).

The research question that Brouwer and Spaninks addressed was, 'is the estimate of willingness to pay in one area (e.g. the area in Friesland) a good approximation of the willingness to pay in the other area (South Holland), and *vice versa*?' They tested two transfer methods: the unit value transfer method and the value function transfer method. The value functions from both studies contained parameters concerning socio-demographic characteristics of the households and their attitudes towards nature conservation in general. With the unit value transfer method, applying the Friesian value (55 florins) to the South Holland site, would underestimate 'true' willingness to pay (74 florins) by 27 %. Conversely, we would overestimate 'true' willingness to pay in Friesland by 36 %.

In this case, the value function transfer method did perform somewhat better. Although it would increase the transfer error from Friesland to South Holland (from 27 % to 28 %) it would reduce the transfer error from South Holland to Friesland (from 36 % to 22 %).

It is perhaps interesting to note that Brouwer and Spaninks did not take the relative scarcity of peat meadow land and its wildlife services into account in their transfer exercise. There are obvious differences between the two sites in this respect: about 40 % of Dutch peat meadow land is in Friesland, a sparsely populated province. The peat meadow land in South Holland is in a densely populated and highly urbanised part of the country. The relative scarcity of the wildlife services between the sites could be an important cause of the difference in willingness to pay for its conservation.

Muthke and Holm-Mueller (2004) test the transferability of contingent valuation estimates for changes in water quality for two German and two Norwegian lakes, thereby testing both national and international transferability. They examine unit value transfer, adjusted unit transfer, and value function transfer using the equivalence testing approach proposed by Kristófersson and Navrud (2005). They find that value transfer between the two German sites produces reasonable results for all transfer methods but especially for the adjusted unit transfer method (transfer error below 20 %). In the adjusted unit transfer method, unit values are adjusted for differences in household income. The study results show very high transfer errors for international value transfer (between Germany and Norway) suggesting that there was insufficient information available to adjust the study site values fully to the policy sites in another country. The authors argue that because economic factors, intrinsic values, tastes and preferences of different cultures and societies show considerable variation, international value transfer can produce large errors. Surprisingly, in the international context the (purchasing power) adjusted unit value transfer method performed worse than the simple unit value transfer method ([5]).

A more prosaic reason for the large international transfer errors in this case may have been that the Norwegian study is itself a bit of an outlier, as Muthke and Holm-Mueller themselves suggest by comparing the Norwegian study to other Norwegian water quality valuation studies. This illustrates the general fact that measurement of transfer errors is itself inexact in that it involves a comparison between transferred values and primary valuation estimates, which are subject to inaccuracies and methodological flaws of their own. In general, primary values are treated as 'true' value observations and transferred values as approximations, whereas they are in fact both approximations.

Muthke and Holm-Mueller also argue that, from a theoretical perspective, the value function transfer method offers the best conditions to generate a valid value transfer. However, it requires good information about the explanatory variables in the value function both at the study and at the policy site, as well as about the coefficients in the primary study. Even if primary studies provide information on the coefficients (which they often do not), data on the explanatory variables in the value function are 'often not available at the policy site, out of date, or not sufficiently precise.' (Muthke and Holm-Mueller, 2004, p. 333).

([5]) On average, the wealthier Germans were willing to pay less than the Norwegians. If a unit value transfer from Germany to Norway (or the other way around) is adjusted for the relatively lower Norwegian income, the transfer error is increased in comparison to the simple unit value transfer.

Kristófersson and Navrud (2007) use identical contingent valuation studies conducted in three countries (Iceland, Norway and Sweden) to examine the validity of value transfers between those countries. The case study estimates use and non-use values for freshwater fish stocks in rivers and lakes. Values are transferred between study sites using both unit transfer and value function transfer. Equivalency analysis is applied to test the validity of value transfers. Use values are generated by recreational fishing, while non-use values are derived from non-angling households' preferences for the preservation of natural fish stocks.

In the case of anglers' use values, unit value transfer between Norway and Sweden produced small transfer errors (< 20 %), while the transfer errors between the Scandinavian countries and Iceland were large (> 100 %). Value function transfer (measuring the usual socio-demographic variables but also fishing activity and expenses) reduced the transfer error with Iceland but not to acceptable levels. Value function transfer actually *increased* the transfer errors between Norway and Sweden. Transfer errors for the non-use values are smaller than for the use values in all cases, except when transferring from Iceland to Norway. For non-use values, transfer errors between Norway and Sweden were 7–8 %, which is very low.

Although they cannot completely explain it in their value function transfer method, the authors suggest that the big differences between Iceland and the Scandinavian countries were due to institutional differences in game fishing, with Iceland having a larger degree of privatisation in recreational fishing and much higher prices of fishing licenses. This, the authors suggest, does not only affect the willingness to pay for recreational fishing, it also seems to affect the non-use values for preserving the Nordic fish stocks.

Brander and Florax (2007) use a meta-analytic value transfer function to estimate values for wetlands in the San Joaquin Valley in California and for the Norfolk Broads in the United Kingdom. These are both internationally renowned wetlands of about the same size (30 000–35 000 ha) for which original valuation studies have been carried out. The meta-analytic value transfer function of Brander and Florax is based on a global database of wetland valuation studies.

By plugging the parameter values of the policy sites into the value transfer function, the authors derive 'transferred' values for ecosystem services of the San Joaquin Valley and Norfolk Broads wetlands, respectively. Brander and Florax compare these 'transferred' values with values from the original valuation studies that were carried out for both wetlands. The lowest transfer error observed in this exercise is 29 % for the valuation of water quality/nutrient retention, recreational hunting and fishing, other recreational activities and amenities in the San Joaquin Valley. Transfer errors of just over 50 % are made for recreational hunting in the San Joaquin Valley, and for biodiversity and landscape maintenance and recreational activities in the Norfolk Broads. The transferred value for bird watching in the San Joaquin Valley, however, is over five times the primary value for this activity.

Lindhjem and Navrud (2008) report on an innovative test of the meta-analytic value function transfer method against variants of the unit value transfer method. They conduct a meta-analysis of contingent valuation results for non-timber forest benefits from Finland, Norway and Sweden. The meta-analytic value function is based on 72 estimates of willingness to pay for non-timber benefits from 26 studies. Comparing the value predictions of their meta-analytic value function to the primary values in their dataset, the authors find mean and median transfer errors of 47 % and 37 % respectively. Lindhjem and Navrud compared this with two variants of the unit transfer method.

In the first variant of the unit value transfer method, for each selected site in their dataset, the mean unit value of all similar studies in the database is calculated, where similarity is expressed in terms of country and other relevant characteristics. In the second variant, the mean unit value is calculated from similar studies from all three countries. Mean unit value transfer from studies from the same country (variant 1) has mean and medium transfer error of 86 % and 41 %. When the mean unit value transfer includes the results of studies from other countries (variant 2) the transfer errors are twice as large (166 % and 85 %).

These results provide some positive support for meta-analytic value function transfer but also suggest that this transfer method is sensitive to meta-model specifications and restrictions. Therefore, the authors argue, we should not cast aside simple approaches (unit value transfer) before we are confident that more complex approaches (meta-analytic value function transfer) perform better, perhaps always. The results also illustrate the difficulty of value transfer between countries, even those with very similar economic, social and institutional characteristics.

The value transfer studies discussed above did not specifically test for the relative transferability of the values of alternative ecosystem services. In this respect it is noteworthy that it has been suggested that the choice experiment method is superior to the contingent valuation method in eliciting preferences for specific ecosystem attributes (i.e. services). In contingent valuation studies it is sometimes the entire bundle of services of an ecosystem that is being valued. Choice experiment studies would therefore lend themselves more easily to value transfer (Rolfe and Windle, 2008) ([6]). A study by Foster and Mourato (2003) confirmed that the choice experiment method is probably superior in valuing individual components of an 'inclusive good' ([7]) than the contingent valuation method. They add, however, that summing up the individual components 'may seriously overestimate the value of the whole set.' If we are interested in estimating the value of the total ecosystem, contingent valuation would be the preferred method (Foster and Mourato, 2003). Because the use of the choice experiment method in ecosystem valuation is of recent origin, a full critique of the method (comparable to the critique of the contingent valuation method) is still lacking.

4.3 Transfer errors

For a number of reasons the application of any of the value transfer methods described above may result in significant transfer errors, i.e. transferred values may differ significantly from the actual value of the ecosystem under consideration. There are three general sources of error in the values estimated using value transfer:

- **Errors associated with estimating the original values at the study site(s)**. Measurement error in primary valuation estimates may result from weak methodologies, unreliable data, analyst errors, and the whole gamut of biases and inaccuracies associated with valuation methods.
- **Errors arising from the transfer of study site values to the policy site**. So-called 'generalisation errors' occur when values for study sites are transferred to policy sites that are different without fully accounting for those differences. Such differences may be in terms of population characteristics (such as income, culture, demographics, education), or environmental or physical characteristics (such as quantity or quality of the good or service, availability of substitutes, accessibility). The magnitude of this type of error is inversely related to the similarity of characteristics of the study and policy sites ([8]). There may also be a temporal source of generalisation error in that preferences and values for ecosystem services may not remain constant over time. Using value transfer to estimate values for ecosystem services under future policy scenarios may therefore entail a degree of uncertainty regarding whether future generations hold the same preferences as current or past generations.
- **Publication selection bias may result in an unrepresentative stock of knowledge on ecosystem values**. Publication selection bias arises when the publication process through which valuation results are disseminated results in an available stock of knowledge that is skewed to certain types of results and that does not meet the information needs of value transfer practitioners. In the economics literature there is generally an editorial preference to publish statistically significant results and novel valuation applications rather than replications. This may result in publication bias resulting in paucity of practically useful data.

There is no clear evidence in research to date of that any of the value transfer methods is superior to the others. There only seems to be some agreement in the literature that the value function transfer method (based on a single study) does not perform very much better than the simpler (adjusted) unit

([6]) Rolfe and Windle (2008) claim that choice experiments 'allow the expression of environmental values as a function of a number of site, population and other characteristics. A choice experiment can be designed in a way so that key elements desired in a benefit transfer function are included in the choice sets as attributes or labels. The choices made by respondents from a survey population thus help to develop a benefit transfer function that can be 'mapped' across to a range of potential policy situations.'

([7]) In our case, the individual components are the individual ecosystem services and the 'inclusive good' is the entire ecosystem.

([8]) In the context of meta-analytic value function transfer, generalisation errors can arise due to the common limitation of meta-analyses to capture differences in the quality and quantity of the services under consideration. It is often the case that the provision of goods and services is indicated in a meta-analysis merely with binary variables, and that quality is not captured at all. This limitation may translate into transfer errors, as the estimated transfer function cannot reflect important quality and quantity differences in characteristics across sites. A similar problem arises where non-identical services have been combined as one explanatory variable in the meta-analysis. Some level of aggregation across service types is often necessary in order to produce a manageable number of variables in the meta-regression, but at the cost of losing specific categories of services.

value transfer method. In practice the usefulness of the value function transfer method is limited by a frequent lack of appropriate data at the policy site. Brouwer and Spaninks argue that if, for example, the value function contains attitude variables that are not routinely recorded by statistical agencies, there is a need for primary data collection of such variables at the policy site. Therefore 'instead of relying upon previous or perhaps outdated contingent valuation (CV) results, one may just as well carry out an original CV study at the policy site' (Brouwer and Spaninks, 1999).

There also seems to be consensus on the relatively poor performance of international value transfer in comparison to domestic value transfer. In some cases, value transfer between similar countries (e.g. Norway and Sweden) is acceptable (for example non-use values of native freshwater fish preservation), in other cases it is problematic (for example non-timber forest benefits). There is not always clear evidence why it is acceptable in one case and not in the other. Value transfer between dissimilar countries is even more problematic (for example angler benefits between Iceland and Norway). In international unit value transfer, adjustment for differences in purchasing power does not always reduce transfer errors.

A limitation of the meta-analytic value function transfer method is related to the reliability of the estimated values. Evidence from the economic valuation literature shows that there are potentially very large transfer errors associated with this approach and that in some cases the relatively simple transfer of unit values may perform at least as well. It is therefore advisable to test the transfer accuracy of a meta-analytic value function in order to provide information about the reliability of the results.

Meta-analytic value function transfer is well suited to valuing large numbers of diverse policy sites because the estimated value function can be applied to a database containing information on ecosystem and socio-economic characteristics of each site. It is a simple operation to enter the characteristics of each policy site into a value function to estimate its value. If the meta-value function is defined in terms of values per unit of area it is also a simple operation to aggregate values over spatial areas. In this case, the approach does not involve aggregation over the affected population but differences in 'market size' can still be taken into account by including the population in the vicinity of the ecosystem as an explanatory variable in the value function.

The value transfer literature has not yet paid much attention to the transferability of the values of individual ecosystem services. It has been suggested that choice experiment results are better suited to value transfer because they focus on individual ecosystem attributes, rather than on often ill-defined 'bundles' of attributes, which are generally the focus of contingent valuation studies. The transfer of values of individual attributes (or services) however leads to the problem of aggregation: how can we sum-up the values of the individual attributes? Aggregation and scaling up is discussed in the next chapter.

5 Scaling up

Scaling up is based on value transfer. Whereas value transfer links a single study site to a single policy site with similar attributes, scaling up transfers values from one or more study sites to a larger geographical setting (see Figure 2.1 above). This seems to leave room for choice, because any mix of primary studies and value transfer methods could be applicable in principle. The question then is how best to deal with this potential in practice.

To start with, Map 5.1 presents the challenge of a scaling-up exercise at the European level, in this case for wetland services values. In 2000, the EU contained more than 50 000 wetlands with a total area of more than 9 million hectares or about 2 % of its land area. The spatial distribution of the wetlands is very skewed with only two countries — Finland and Sweden — containing over two-thirds of the total number of wetlands and half of the total area.

As a basis for the valuation of these wetlands, we identified 51 European wetland valuation publications from the period 1988–2008. The publications contain 90 observations on the value of a service at a specific site. There is little spatial correlation between the location of wetlands and wetland valuation studies. There is little or no value information on large wetland areas in, for example, Hungary, Ireland, Romania and Scandinavia.

It should be clear that the attempt to assess the value of historical or projected changes of wetland services across Europe on the basis of this small set of primary valuation studies poses some major challenges. The two major problems in scaling up values concern information and aggregation.

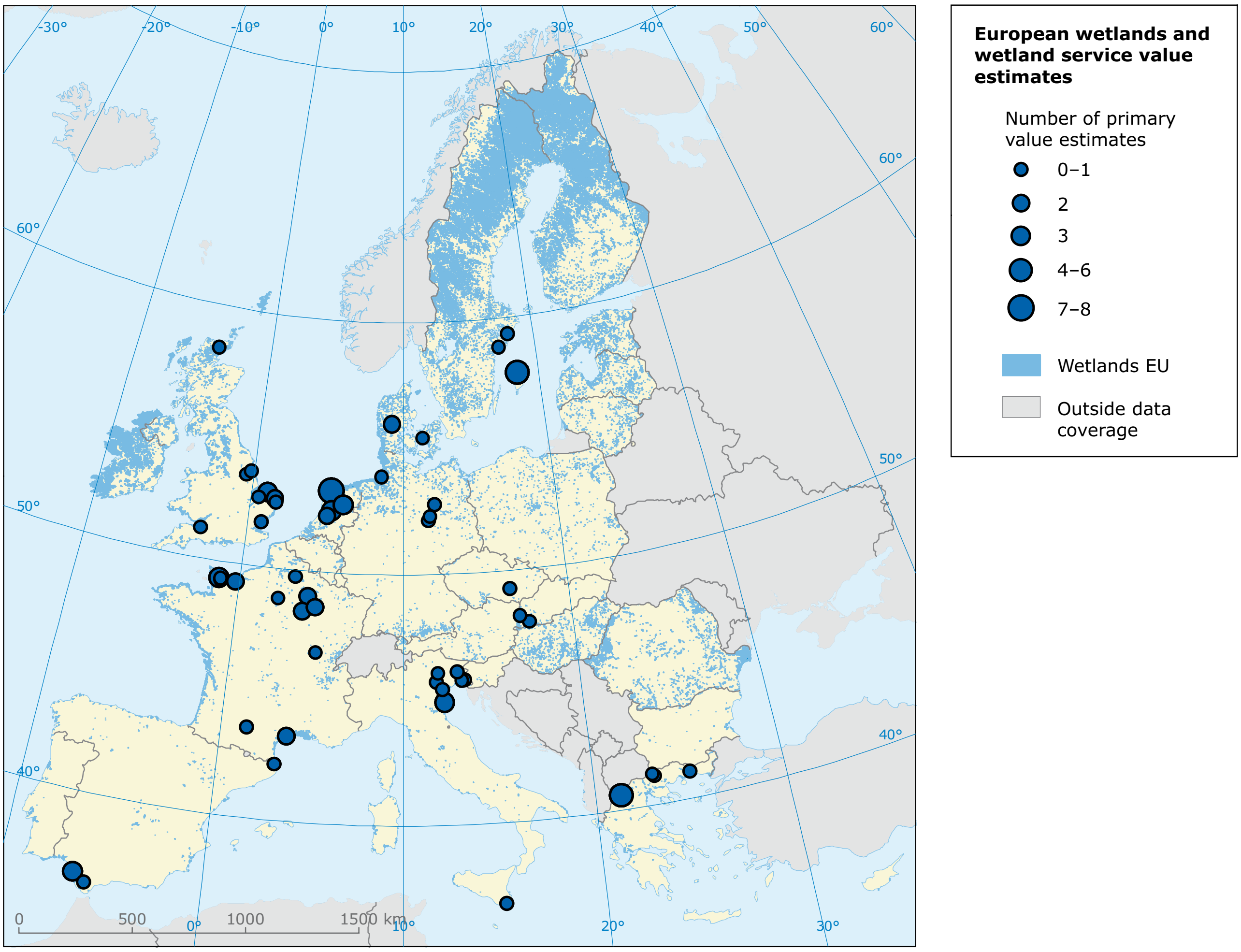
5.1 Information

Information from primary studies is often scarce, fragmented, incomplete and of varying quality. Information about relevant characteristics of ecosystems and their beneficiaries at the policy sites can be hard to find or simply unavailable.

One of the key questions in value transfer and scaling up is the choice of the unit of transfer. Different studies can measure and report values in different dimensions: value per household or individual per month, per year, or some present value (one-time donation), value per recreational trip, value per activity day, value per unit of area. And all of these values can be expressed in different currencies in different years. Hence, some form of standardisation is often necessary, and can either be the same across all services or specific for each service. Standardisation to a common unit of transfer is a non-trivial step in scaling up.

Consider the case, for example, of a primary valuation study that has measured the recreational value per visit to an ecosystem site. The best way to transfer this value to the policy sites in the larger geographical region would be to determine the value per visit and to multiply it by the number of visits at the policy sites (perhaps adjusting for socio-economic differences between visitor populations). Let us assume, however, that the number of visits to the policy sites in the larger geographical region is unknown. The second best unit of transfer would perhaps be a per area value. If the areas and number of visits to the policy sites were very different from those of the study site, a simple unit transfer would result in a potentially large transfer error. The adjusted unit transfer and meta-analytic value function transfer methods can adjust for observed differences in areas and visits (perhaps via population density as a proxy for visitor rates). These adjustments would likely reduce the transfer error, but would not eliminate it and the error would probably still be greater than the error of the best (per visit) unit of transfer *if* that had been feasible. This illustrates that choosing the unit of transfer requires a careful deliberation taking into account information from the primary studies and the available information regarding the policy sites. As a rule, the unit of transfer should be as close as possible to the original value unit in the primary valuation study.

Map 5.1 Spatial distribution of wetland value estimates and wetlands across Europe



Source: IVM, 2010.

Reliable value transfer has to account for contextual differences that explain values for the study and policy site. There are two different sets of spatial attributes to be addressed in economic valuation of any environmental change:

- the spatial pattern of the social, demographic and psychological characteristics of the affected population;
- the spatial variance in physical characteristics of the goods and services under valuation.

Assuming that population preferences are constant over space ignores demographic, socio-economic and cultural differences between regions, or the influence of location and distance on environmental values. Ignoring the spatial variance in physical characteristics of goods and services implies an assumption that they are constant in terms of quantity and quality.

Geographic information systems can be used to help link valuation data with information on the physical characteristics (such as ecosystem size, availability of substitute sites) and socio-economic characteristics (income, population, education) of the policy site.

5.2 Aggregation

The second major problem for scaling up exercises is that primary studies have usually assessed the values of ecosystem services in isolation. That is, they have assessed the value of particular services under the assumption that all else would remain equal. As already noted in Chapter 2, at the margin a small change in ecosystem service provision (e.g. the loss of a small area) will not affect the value of services from other ecosystem sites. Non-marginal changes in ecosystem service provision, however, will affect the value of services from the remaining stock of ecosystems. As the ecosystem service becomes scarcer, its marginal and average values will tend to increase. This means that simply multiplying a constant per unit value by the total quantity of ecosystem service provision is likely to underestimate total value of a negative change.

One of the most important contextual factors in a value transfer exercise is the availability of substitutes. Ignoring substitutes means that if the transfer is performed between a landscape poor in ecosystem services to a landscape rich in ecosystem services, marginal values are likely to be overestimated (Bateman *et al*., 1999). The question is what happens to the willingness to pay for one good if the quality of a comparable, substitute good increases. A substitution effect in economics is usually defined as the increase of demand for good A when the price of good B increases. Substitutes and complements can be take the form of different services at one ecosystem site or identical services at spatially separate ecosystem sites.

The consequence of disregarding substitutes is generally an overestimation of willingness-to-pay, as the sum of the value of goods measured individually is higher than the value measured for all goods at once. For instance, respondents in an area with several lakes whose water quality is polluted will value cleaning up the first lake more than cleaning up the second lake, because first the first lake can be a substitute for the second lake, and second the respondent has a limited budget, which reduces the money available for cleaning up the second lake. Valuing goods separately and then adding up the values will overstate the true value, as every respondent will treat the ecosystem under study as if it were the first good. As distance from the site or the geographical scale of the study increases, the number of substitutes is likely to increase.

Disregarding complementary sites causes underestimation of willingness to pay. Complementarity occurs when goods are consumed jointly, for instance when two sites are visited during the same trip, or when there are synergy-effects in production, for instance when quality increases at one site automatically increase the quality of another site due to dependent ecosystems. The value of one site is therefore likely to be dependent on other available alternatives and their characteristics.

An important factor in a scaling-up study is therefore to determine the relevant substitutes for a certain ecosystem or ecosystem service. Different criteria have been used to determine the relevant alternatives, specifically:

- all available similar ecosystems in the study area or within a certain range;
- all similar ecosystems known or visited by the respondent;
- all nature sites in the study area;
- all possible recreation areas (not necessarily nature based).

Aggregation can also refer to summing up the values of different ecosystem services of the same ecosystem. This approach may lead to double counting. As long as the functions are entirely independent, adding up the values is possible. However, ecosystem functions can be mutually exclusive, interacting or integral (Turner *et al*., 2004). The excludability or interaction of ecosystem functions and values can also be dependent on their relative geographical position, for instance with substitutes that are spatially dependent.

5.3 Non-constancy of marginal values and critical thresholds

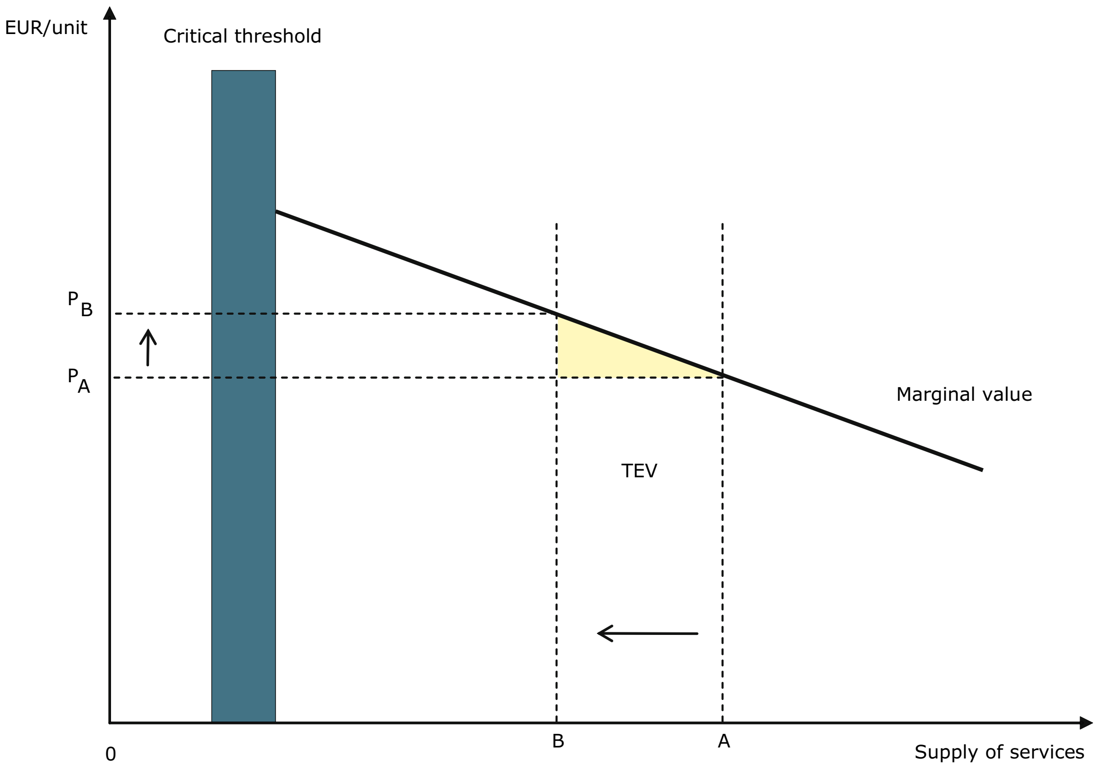
Conceptually, the economic value of losing the provision of an ecosystem service can be expressed as the area under the social demand curve for the service that is bounded by the pre-change level of provision and the post-change level of provision, everything else being equal (as presented in Figure 5.1).

Figure 5.1 shows a downward sloping demand curve for the flow (or supply) of ecosystem services. The total economic value of a small loss of services (from supply A to supply B) can be evaluated as the area under the demand curve. In the diagram above, the marginal unit value of the services increases from P_A to P_B when the supply decreases from A to B. Assessing the total value of this change using only the marginal unit value at supply A (P_A) would result in an underestimation of this value. The magnitude of this underestimation error is measured by the dashed triangle. Because of this it is necessary to account for the change in marginal value over the extent of the change in service provision.

Changes in service provision can be assessed until a critical ecological threshold is reached (vertical bar in Figure 5.1), from which point onwards it is no longer possible to obtain meaningful economic values. A critical threshold is usually understood to be the point at which an ecosystem ceases to function. It is of course difficult to exactly define critical thresholds for every ecosystem service but the general idea is appealing.

A slightly different way of making the same point is that valuation studies always measure willingness to pay for ecosystem services around present levels of overall provision (studies usually focus on one site, with the implicit or explicit assumption that the level of provision of services from substitute sites is not changed). Large changes in the overall level of provision are beyond the domain of our observations and are therefore principally unknown. This would make the assessment of the value of a complete loss of an ecosystem service (e.g. from supply level B to supply level 0) impossible. Note the assumption of critical thresholds will be more relevant for some services (like biodiversity and some locally important regulating services) than for others (for example, the role of European forests in the entire global carbon cycle will always remain 'marginal').

Figure 5.1 Valuing changes in the provision of ecosystem services



Source: Adapted from Turner *et al.*, 2003.

6 Case study: wetlands at the national level

The case study presented here assesses the economic value of changes in the provision of wetland services in the Netherlands and the Baltic states between the years 2000 and 2006. Corine Land Cover maps were used to assess the land use changes that took place. These land-use changes are appraised by two of the value transfer methods described in Chapter 4: the unit transfer method and the meta-analytic value function transfer method. Moreover, the case study explores whether combinations of transfer methods and available primary study results can be used in scaling-up applications.

The case study illustrates how scaling-up methods can be applied in practice. Because of uncertainties in the determination of the historical land use change on the basis of the Corine Land Cover maps, the quantitative results of the case study should be interpreted with caution.

The chapter starts with a description of the wetlands in the case study countries and reviews the primary valuation studies that have been carried out there.

6.1 Wetland availability and change

Under the Ramsar convention, Estonia has 12 sites designated as wetlands of international importance, Latvia has six and Lithuania five. Compared to the total area of the country, the area of the wetlands is respectively 5 % in Estonia, 2.3 % in Latvia and less than 1 % in Lithuania.

The Baltic States border the Baltic Sea and therefore host many salt marshes and intertidal mudflats. The main threats to these wetlands are eutrophication and toxic substances (Wulff *et al*., 2001) but other threats include the invasion of exotic species and other human influences, such as tourism and fishing. This has led to large algal blooms and the disappearance of top predators, for example eagles and seals (Wulff *et al*., 2001)

Among the inland wetlands, there are many peat bogs, such as the Endla Nature Reserve and the Soomaa National Park in Estonia, the Lubana Wetland Complex and Teici and Pelecares bogs in Latvia, and Cepkeliai in Lithuania. Peat bogs play a very important role in regulating atmospheric greenhouse gases (Chmura *et al*., 2003): they are net sinks for CO_2 and potential sources of methane (CH_4).

Some of the inland wetlands are under threat due to eutrophication and insufficient treatment of sewage water. Other river and delta areas, such as the Nemunas Delta, also suffer from hydromorphological changes, such as dams. In addition, continued drainage for agricultural and forestry purposes remains an important threat. Sometimes, for instance in Estonia, peat is still excavated for fuel.

Compared to the Baltic States, the Netherlands has a much larger number and proportion of total area that fall under the Ramsar Convention. Almost 20 % of total land area is designated as internationally important wetland. The Wadden Sea, the Wadden Islands and North Sea coast, the IJsselmeer and the Delta in the province of Zeeland are the largest wetlands. Except for the IJsselmeer, these large wetlands are all saline or brackish waters, some of them with tidal mudplains. In addition to their natural amenity values, the large water bodies, e.g. the Wadden Sea and the IJsselmeer, also have great importance for fisheries, recreation and shipping.

Wetlands in the Netherlands are extremely important to bird life, as many west European water birds hibernate or breed there. Much of the country's flora and fauna depends on wetlands. Furthermore, wetlands play an important role in water purification, retention and flood storage (Janssen *et al*., 2005).

Although the remaining peat bogs and fens are relatively small, they form a cultural aspect of the landscape. Wetlands also have direct use values in terms of recreation. Most of the outdoor recreation in the Netherlands takes place at beach sites and water bodies. However, the intense use also poses a threat to ecosystem health. For instance, the Oostelijke Vechtplassen, which comprises a large area of shallow lakes, canals, fenland and reedbeds, provides a habitat for a large number of endangered species of insects, birds, mammals, plants and mosses. The area is not only used for tourism and leisure, but also for commercial fisheries, farming and boating. Discharges from farmland and households, threaten the ecosystem, as do reed harvesting and pleasure navigation. The inland wetlands face even greater threats from water abstraction for agriculture (Goosen and Vellinga, 2004). Lowering groundwater tables increases agricultural production but also results in soil subsidence. Large hydromorphological changes (dams, weirs), eutrophication and toxicity are further threats throughout the river basins.

Table 6.1 provides an overview of the wetlands in the Netherlands and the Baltic States as included in the Corine database for the year 2000. Four types of wetlands are identified in the Netherlands: inland marshes, peat bogs, salt marshes and intertidal mudflats. For Lithuania and Latvia, the database only includes inland marshes and peat bogs. For Estonia, the database also includes one salt marsh.

Wetland change over the period 2000–2006 has been assessed on the basis of Corine Land Cover maps. Net changes in wetland area were found to be positive in the Netherlands (+ 1 459 ha), Estonia (+ 611 ha) and Lithuania (+ 47 ha), and negative in Latvia (– 178 ha). These changes are small in relation to total area (see Table 6.1).

We found no primary valuation studies on wetland services from any of the Baltic States. For the Netherlands we identified six publications that appraised one or more wetland services from different wetlands. Three publications assessed various ecosystem services of the Wadden Sea. De Groot (1992) appraised the economic benefits of flood prevention, storage and recycling of human waste, nursery provision, aquaculture and recreation, food production, and services to education and science. He used a mix of different valuation methods, including cost approaches and market price methods. Spaninks *et al*. (1996) carried out a contingent valuation survey to estimate the willingness-to-pay of the Dutch population to attain natural conditions in the Wadden Sea and to protect biodiversity. De Blaeij *et al.* (2004) assessed recreational (bird-watching) benefits in a choice experiment. Services of other wetlands were appraised by De Groot *et al*. (1998), Bos and van den Bergh (2002), and Hein *et al*. (2006). The studies valued different (sets of) services and used different valuation methods.

6.2 Unit value transfer

The Netherlands

The primary valuation studies discussed above cover four wetland types (inland marsh, peat bog, salt marsh and intertidal mudflat) and a range of ecosystem services (including recreation, biodiversity, habitat provision, materials, and improvements to water quality). The studies provide us with 15 separate value estimates for combinations of wetland type and ecosystem service (see Table 6.1). The unit values across the studies differ: while the unit value in some studies is willingness to pay per household, other unit values include total observed market transactions and area-based units. Lacking one single preferred value unit from the primary studies, and considering that per hectare values are easy to use when valuing area

Table 6.1 Area of wetland types in Netherlands and Baltic States in hectares (ha)

	Netherlands	Lithuania	Latvia	Estonia
Inland marshes (ha)	33 910	18 998	23 274	76 913
Peat bogs (ha)	7 728	39 214	134 212	123 070
Salt marshes (ha)	9 368	0	0	416
Intertidal mudflats (ha)	228 885	0	0	0
Total (ha)	**279 891**	**58 212**	**157 486**	**200 399**

Source: Corine, 2000.

changes, we converted all value estimates into per hectare values in constant 2005 euro. We use these Netherlands unit values to estimate the value of the change in wetland stock by multiplying them by the change in area of the relevant wetland type in the country (Table 6.2).

The Dutch primary studies cover fewer services than potentially available according to the classification of wetland services that was presented in Table 3.1. Lacking are, for example, flood and storm protection services, water supply services, and climate stabilisation and climate change mitigation services. It is not *a priori* known whether these services are not very important in the Dutch situation or whether there are other reasons why these services have not been valued. But there is also not enough evidence to assume that the absent services have no value. Furthermore, there are also differences in service coverage between wetland types (intertidal mudflats and salt marshes have a greater coverage than inland marshes and peat bogs). Finally, the distinction between recreation and biodiversity services is not always clear in primary studies, so we have included in Table 6.2 a combined service: recreation and biodiversity. This somewhat patchy coverage of wetland services and types is probably typical of most sets of primary valuation studies.

The unit value transfer method is a relatively quick and easy transfer method. The reliability of its results depends on the quality of the primary studies, the coverage of services and ecosystem types in the set of primary studies, the extent to which the study sites and the policy sites are comparable (and hence unit value transfer is appropriate) and the extent to which substitution and income effects and demographic changes may be neglected. With good primary studies, unit value transfer may be quite appropriate for domestic scaling up for a small change in ecosystem service provision. Notice that in domestic scaling up through the unit value transfer method the results of the primary studies remain preserved. For example, changes in the supply of services of the Wadden Sea ecosystem are valued by studies that have specifically focused on the Wadden Sea. Depending on the quality of the primary studies, this gives some confidence in the scaling-up results. It is also noteworthy, however, that a potential weak point of the scaling-up exercise is the incomplete coverage of ecosystem services in the set of primary studies.

The Baltic States

Due to the lack of primary valuation studies on wetland services in the Baltic States, it was not possible to apply the unit value transfer method to estimate value of changes in wetland area in the Baltic States. In principle, however, such an appraisal would be possible using the adjusted unit value transfer method. A database of wetland valuation studies could be used to choose value observations from the most similar sites, where 'similarity' could be defined in terms of wetland type, services provided, or any other combination of attributes. For the transfer to the Baltic States, the unit values could be adjusted to account of, for example, differences in income, population density, wetland abundance and wetland size. The adjustment factors (elasticities) could be derived from the meta-analytic value function to which we now turn.

Table 6.2 Annual value of (net) change in wetland area in the Netherlands in the period 2001–2006 per wetland type and wetland service (euro)

	Recreation	Biodiversity	Recreation and biodiversity	Habitat	Water quality	Materials	Total
Inland marshes	820 000						820 000
Peat bogs	21 000	27 000		2 000		3 000	52 000
Salt marshes	– 1 000	– 2 000	– 5 000	– 1 000	– 44 000		– 53 000
Intertidal mudflats	42 000	129 000	298 000	80 000	2 554 000		3 102 000
Total	**882 000**	**153 000**	**292 000**	**80 000**	**2511**	**3 000**	**3 921 000**

Source: Corine, 2000.

6.3 Meta-analytic value function transfer

An alternative method for scaling up makes use of a meta-analytic value function. The meta-analytic value function for temperate wetlands is presented in Table 6.3 and is described in full in Brander *et al*. (2008). The meta-analytic value function relates willingness-to-pay per hectare of wetland to a number of explanatory variables, including valuation method, wetland type, size, wetland services and some spatial context variables such as income per capita, population density in the vicinity of the wetland and a measure of regional wetland scarcity. Note that this meta-analytic value function is based on available data for temperate wetlands globally, i.e. it is estimated using value data from the Australia, Canada and the United States, as well as Europe.

We use the meta-analytic value function to assign per-hectare values to all wetlands in 2000 and 2006, evaluating them at 2006 population densities and per capita incomes, and taking account of the differences in wetland scarcity and wetland size between these years. For this purpose, the Corine Land Cover maps were overlaid with socio-economic maps that report regional data on

Table 6.3 Meta-analytic value function for temperate wetlands

	Variable	Coefficient	p-value
	(constant)	– 3.078	0.187
Study variables	Contingent valuation methods	00.065	0.919
	Hedonic pricing	– 3.286***	0.006
	Travel cost method	– 0.974	0.112
	Replacement cost	– 0.766	0.212
	Net factor income	– 0.215	0.706
	Production function	– 0.443	0.523
	Market prices	– 0.521	0.317
	Opportunity cost	– 1.889**	0.035
	Choice experiment	00.452	0.635
	Marginal	01.195***	0.008
Wetland variables	Inland marshes	00.114	0.830
	Peat bogs	– 1.356**	0.014
	Salt marshes	00.143	0.778
	Intertidal mudflats	00.110	0.821
	Wetland size	– 0.297***	0.000
	Flood control and storm buffering	01.102**	0.017
	Surface and groundwater supply	00.009	0.984
	Water quality improvement	00.893*	0.064
	Commercial fishing and hunting	– 0.040	0.915
	Recreational hunting	– 1.289***	0.004
	Recreational fishing	– 0.288	0.497
	Harvesting of natural materials	– 0.554	0.165
	Fuel wood	– 1.409**	0.029
	Non-consumptive recreation	00.340	0.420
	Amenity and aesthetics	00.752	0.136
	Biodiversity	00.917*	0.053
Context variables	GDP per capita	00.468***	0.001
	Population in 50km radius	00.579***	0.000
	Wetland area in 50km radius	– 0.023	0.583

Note: OLS results: R2 = 0.49; Adj. R2 = 0.43.

'***' denotes 1 % statistical significance; '**' denotes 5 % statistical significance; '*' denotes 10 % statistical significance.

GDP per capita and population. We then multiply the change in area of each wetland with the *average* of the 2000 and 2006 per hectare value for each wetland.

The Netherlands

For the purposes of illustration, Table 6.4 shows the values of key variables used in calculating the value of change in a single wetland in the Netherlands (the Loosdrecht Lakes), about 30 km south-east of Amsterdam. Over the period 2000–2006, the wetland reduced in size from 1 614 ha to 1 529 (a change of 85 ha). The abundance of wetlands within a 50 km radius of the wetland also declined slightly over the same period. These values, together with the population within a 50 km radius and the GDP per capita for the NUTS3 region in 2006, are plugged into the meta-analytic value function to estimate per hectare values for the wetland in its 2000 'state' and its 2006 'state'. The changes in wetland area and abundance cause the value per hectare of the wetland to increase slightly from 5 355 to 5 444 euro/ha (the average value per hectare between the two states is therefore 5 400 euro/ha). Multiplying the average value by the change in area gives the value of the change. That totalled a loss of 459 938 euro in 2006 compared with 2000.

Table 6.5 presents the results of the calculations for all wetland changes in the Netherlands over the period 2000–2006. For comparison, Table 6.5 also shows the results of the calculations using the unit value transfer method.

These calculations with the meta-analytic value function transfer method suggest that the welfare gains and losses due to wetland change in the period 2000–2006 have more or less cancelled each other out. On balance, the calculations show a net welfare loss (– 186 158 euro). As we saw earlier, calculation with the unit value transfer method suggests a small welfare gain (+ 3 921 112 euro).

Table 6.4 Example of the data used in calculating an individual wetland value in the Netherlands (Loosdrecht Lakes)

Variable	Total
Wetland area in 2000 (ha)	1 614
Wetland area in 2006 (ha)	1 529
Wetland abundance in 2000 (ha)	8 795
Wetland abundance in 2006 (ha)	8 755
Population in 2006	6 330 324
GDP per capita (2006 USD PPP)	27 582
Value per hectare 2000 (2005 EUR)	5 355
Value per hectare 2006 (2005 EUR)	5 444
Average per hectare value (2005 EUR)	5 400
Change in wetland area (ha)	– 85
Value of change (2005 EUR)	– 459 938

Table 6.5 Volumes and values of wetland change in the Netherlands, 2000–2006, using the meta-analytic value function transfer method and the unit value transfer method

	Meta-analytic value function transfer	Unit value transfer
Loss in area (ha)	– 9 044	– 9 044
Gain in area (ha)	+ 10 503	+ 10 503
Net change in area (ha)	+ 1 459	+ 1 459
Welfare loss (euro)	– 28 604 015	– 18 935 152
Welfare gain (euro)	+ 28 417 856	+ 22 856 263
Net change in welfare (euro)	– 186 158	+ 3 921 112

Using the unit value transfer approach has the advantage of allowing the estimation of service-specific values. This is limited, however, to those services for which unit values are available. In the current example, available primary wetland valuation studies for the Netherlands only cover a subset of ecosystem services. This could explain why the unit value transfer method estimated lower aggregate values (in absolute terms) for both gains and losses.

With the unit value transfer method, the unit values remain constant in the analysis and are not adjusted to reflect changes in the characteristics of the wetland stock over time (i.e. there is no adjustment in value for changes in the size of individual wetland sites or for changes in the availability of substitutes). The meta-analytic value function transfer approach on the other hand, enables us to estimate wetland values reflecting all ecosystem services and adjust for changes in wetland characteristics. The meta-analytic value function transfer method values Netherlands wetland losses as greater than gains. The main reason for this is that the mean per hectare value of the wetlands that decreased in size was higher than the mean value of the wetlands that increased in size. Hence, the loss because of the decrease in the area of valuable wetlands could not be totally offset by the increase in the area of less valuable wetlands ([9]).

The Baltic States

Table 6.6 presents the results of calculating the change in wetland area in the Baltic States using the meta-analytic value function transfer method.

6.4 Combinations of value transfer methods

An exploration of whether combinations of transfer methods and available primary study results can be used in scaling-up applications suggests that there are various possibilities.

First, it is good practice to cross-check results derived by different methods. In the discussion of value transfer methods (Chapter 4), it was concluded that no single transfer method is superior to other methods on all counts. Therefore, confidence in the transfer will increase if the results of alternative methods are not too dissimilar.

For the Dutch case study, a simple check is the difference in average per hectare wetland value between the unit value and the meta-analytic value function transfer method. Using the numbers in Table 6.5, the average values per hectare wetland gain in the Netherlands are EUR 2 706 per ha using the meta-analytic value function transfer method and EUR 2 176 per hectare using the unit value transfer method, respectively. This is a small difference, especially when one considers the incomplete coverage of services by the unit value transfer method. It would also be good practice to cross-check, if possible, the per hectare values against other metrics, e.g. per household willingness to pay for wetland conservation.

Second, different transfer methods can be used for different services, provided that the services are (sufficiently) independent. Strictly speaking, independency of services is a rather strong condition

Table 6.6 Volumes and values of wetland change in Estonia, Latvia, and Lithuania, 2001–2006, using the meta-analytic value function transfer method

	Estonia	Lithuania	Latvia
Loss in area (ha)	– 6 222	– 348	– 479
Gain in area (ha)	+ 6 834	+ 396	+ 302
Net change in area (ha)	+ 612	+ 48	– 178
Welfare loss (euro)	–378 224	– 67 319	–79 362
Welfare gain (euro)	+ 281 507	+ 166 100	+ 86 155
Net change in welfare (euro)	– 96 717	+ 98 781	+ 6 793

([9]) A contributing factor is the diminishing returns to size as explained in the example of the Loosdrecht Lakes. This factor is small in comparison to the main factor: the difference in mean per hectare values.

but one might for example argue that global and local services of ecosystems are sufficiently independent. The values of local services (recreation, amenity, water quality) could then be transferred by geo-specific transfer methods such as the meta-analytic value function transfer method described in this report, while the global service values (e.g. carbon sequestration or 'global' non-use values of very unique ecosystems) could be added to the total economic value by simple unit value transfers.

Third, a distinction could be made between well-studied 'exceptional' or 'hotspot' ecosystems and the more mundane ecosystems with local significance. For example, Map 5.1 suggests that the Wadden Sea in the Netherlands and the Norfolk Broads in the United Kingdom are well-studied wetlands with international appeal and significance. A scaling-up exercise could simply value the services of these 'hotspot' wetlands with the 'best' primary value estimate (or some average estimate across a number of studies), and apply a common value transfer method to the other, more commonplace wetlands. The situation changes when no primary valuation studies are available for 'hotspot' wetlands in a scaling up exercise. Applying a unit transfer may seriously underestimate the values of these sites. If applying a (meta) function transfer, outlying values will play a smaller role in the overall outcome, the larger the geographical scale of the scaling-up exercise.

7 Discussion of policy applications

Various guidelines are available for value transfer (for example Navrud, 2007; Eftec, 2009). There is, however, little evidence of applying scaling-up methods in the literature and no guidelines on this approach. The present chapter aims to provide some guidance by discussing the results with a view to policy applications.

Scaling up is the use of existing data on economic values of ecosystem services for an assessment of these values at a larger geographical scale. In a scaling-up exercise, a number of questions have to be addressed and answered. The questions relate to:

- policy context;
- scientific knowledge base;
- primary valuation data;
- transfer methods and units of transfer;
- spatial data and other data at the target geographical scale;
- aggregation and scaling up;
- transfer errors and uncertainty.

7.1 The policy context

For which policy decisions is scaling up needed? It is important to realise that scaling up (and appraisal of ecosystem services in general) is only relevant in the evaluation of relatively small changes in ecosystem service provision. In the past, there have been attempts to attach economic values to total global ecosystems but this has no economic significance. No scaling-up exercise will ever be able to answer a question like, 'what is the value of all wetlands in Europe?' Scaling up *may* help in answering a question like 'what is the benefit of halting wetland loss in Europe in comparison to a trend of continuing wetland loss over the next twenty years?'

The policy problem is also of importance in determining the maximum acceptable (transfer) error in the final appraisal. Cost-benefit analyses of particular policy options or damage assessments to be used in court require a higher level of accuracy and detail than broad impact assessments of proposed policies or regulations, or studies that serve generally to underline the need for policy action, to prioritise between different policies (cost of inaction studies) or to raise awareness.

7.2 Scientific knowledge base

Economic valuation studies and scaling up cannot fill the gap when scientific knowledge is lacking. If, in a specific area, there is a lack of scientific knowledge about important relationships between environmental pressures, ecosystem functioning and the provision of ecosystem services, economic valuation will not add anything to our understanding of these relationships. Nor can economic valuation in such a situation appraise policies that are directed at ecosystem conservation.

Recent value transfer guidelines contend that: '*One of the major challenges for practical benefits transfer is to ensure, from the outset, that the change in provision is understood and quantified. It is clearly unreasonable to expect either primary valuation studies or benefits transfer to derive robust values for a good when the quantity change in provision (and/or the quality change) is unknown. In the case of environmental goods, prediction of the quantity change in provision typically requires a prior basis of natural science*.' (Eftec, 2009).

7.3 Primary valuation data

With respect to primary valuation data, there are two questions to be answered. First, what exactly are we looking for (the definition of the good to be valued)? Second, where should we look?

It is important that a clear definition of the good (service) to be valued is adopted at the start of the scaling-up exercise. As was briefly discussed in Chapter 3 of this report, there is some ambiguity in the classification of ecosystem services in general and in ecosystem valuation studies in particular. If one wants, for example, to appraise the benefit of a policy halting the loss of wetlands in Europe, one should be sure about the definition of wetlands, and about the exact services that are provided by wetlands. It is commonplace to talk about the 'biodiversity' value of ecosystems but what is it exactly? Is it a non-use value, an option value, is it (also) an element of recreational use values? How do we measure its change? These are difficult questions but they should somehow be addressed.

Good starting points for locating primary valuation studies are databases such as the Environmental Valuation Reference Inventory (EVRI) (www.evri.ca), the Nature Valuation and Financing (NV&F) database (www.eyes4earth.org/casebase), and ENVALUE (www.environment.nsw.gov.au/envalue/). Such databases have often been established to support researchers and policy advisers in value transfer or scaling-up exercises. Other sources are academic journals (e.g. *Ecological Economics*, *Environmental and Resource Economics*, *Journal of Environmental Economics and Management*, *Land Economics*) and experts. In addition, TEEB is developing a database containing values of ecosystems services as a data source for benefit transfer operations.

After collecting valuation studies of interest, it is good to review the studies in terms of scientific soundness, relevance and richness in detail (for specific guidelines on quality assurance: see Desvousges *et al*., 1998; Söderqvist and Soutukorva, 2009). At this stage, a 'gap' analysis is a useful tool to identify the main gaps in the primary valuation literature in terms of services and regions. Based on this gap analysis, the analyst can adjust her search for primary valuation studies in the direction of the gaps, or decide to carry out (or ask funding for carrying out) additional primary valuation studies. In the end, when the availability of primary data is too small for a scaling up exercise — based on criteria to be developed — any value transfer method may lead to unacceptable transfer errors, and hence scaling up is not the way to go. Primary research will then be necessary for a reliable outcome.

Unique wetlands with services that exceed instrumental values should, of course, receive a separate treatment in a policy decision process (if not already singled out on the basis of biodiversity and ecosystems protection policies).

7.4 Transfer methods and units of transfer

Chapter 4 of this report provides a description, discussion and appraisal of the four value transfer methods.

- Unit value transfer
- Adjusted unit value transfer
- Value function transfer
- Meta-analytic value function transfer.

Further analysis of these methods is available in the specialised literature (e.g. Navrud and Ready, 2007).

Having reviewed tests of these four methods, it was concluded that there is no clear evidence in current research that any the value transfer methods is inherently superior. There is only some agreement in the literature that the value function transfer method (based on a single study) does not function very much better than the simpler (adjusted) unit value transfer method.

In a scaling-up exercise different transfer methods might be used for different services, provided that the services are sufficiently independent. The values of local services (recreation, amenity, water quality) could then be transferred by geo-specific transfer methods such as the meta-analytic value function transfer method described in this report, while global service values (e.g. carbon sequestration, 'global' non-use values of very unique ecosystems) could be added to the total economic value by simple unit value transfers.

In a scaling-up exercise, it might also be possible to make a distinction between 'exceptional' or 'hotspot' ecosystems and ecosystems with local significance. The values of these 'hotspot' ecosystems (e.g. Wadden Sea, Norfolk Broads) are often well studied, and their 'best' primary estimates could be preserved in the scaling-up exercise, while applying value transfer methods to the other, more commonplace ecosystems.

The *unit of transfer* in a scaling-up exercise is an important choice variable. As a rule, the unit of transfer should be as close as possible to the original value unit in the primary valuation study. Hence if the recreational value of an ecosystem is measured in terms of willingness to pay per visit, a transfer of this value is preferred. If it is not possible to use the original unit (because different primary studies used different units, or because reliable quantity information (number of visits) is not available at the policy sites), some transformation of the original values is necessary in the standardisation process. Such transformations, if applied uncritically, can be a potentially large source of transfer error and should therefore always be critically checked if possible. This issue is addressed further in Section 7.7 below.

7.5 Spatial data and other data at the target geographical scale

There is a close connection between the choice of a value transfer method and the data needs at the target geographical scale. With the simplest value transfer method, data are needed on the value unit in the policy area: e.g. number of recreational visits to a particular ecosystem. If meta-analytic value function transfer is used, the data requirements may be substantial (depending on the number of explanatory site, service, and context-specific explanatory variables in the meta-analytic regression). In actual scaling-up exercises, the choice of transfer method and the level of detail will be a compromise between what is desirable and what is possible.

Geographic Information System (GIS) tools are well-suited to present the economic ecosystem service values collected in relation to ecosystem characteristics, population and other socio-economic data. Local differences in the abundance or scarcity of ecosystem services and other spatial issues such as distance-decay effects can be illustrated and further evaluated. The combination of economic and spatial analysis facilitates the appraisal of (expected) changes in land cover and land use and thus in the provision of ecosystem services at the (scaled up) policy sites. Moreover, GIS-based models can serve as a tool to facilitate the actual scaling-up exercise. They may assist both researchers and decision-makers to estimate the similarity of (a) specific policy site(s) to the characteristics of the original study site(s). Another advantage of most GIS tools is that they can perform spatial analysis at local, regional and global levels.

7.6 Aggregation and scaling up

Scaling up provides the possibility to combine (several sets of) primary data and one or more value transfer methods to assess the economic value of changes in ecosystems services at a larger spatial scale. The magnitude of the change under study affects the direct applicability of values taken from primary research.

Primary valuation studies have usually assessed the values of ecosystem services in isolation, that is, they have assessed the value of particular services under the assumption that all else would remain equal. At the margin, a small change in ecosystem service provision (e.g. the loss of a small area) will not affect the value of services from other ecosystem sites. Non-marginal changes in ecosystem service provision, however, will affect the value of services from the remaining stock of ecosystems. As the ecosystem service becomes scarcer, its marginal and average values will tend to increase.

It is important in a scaling-up exercise to take account, to the extent possible, of cross-substitution effects between ecosystem services and diminishing returns to scale. The present report does not offer general guidance on how to do this in all situations. However, the case study on scaling up wetland service values presents an approach, based on a meta-analytic value function with coefficients for wetland size, wetland scarcity, per capita income and population density.

7.7 Transfer errors and uncertainty

Value transfer and scaling up can generate substantial transfer errors. These errors may be limited by carefully addressing potential measurement and generalisation errors and publication biases, but they can never be totally avoided. At a more fundamental level it can be argued that both the primary and the transferred values are estimates subject to sampling error, so there is no 'certain' benchmark from which to measure transfer error (Brander and Florax, 2007). Nevertheless, based on the sample of available primary studies the analyst is advised to carry out and report on 'within sample' and 'out of sample' tests (e.g. Lindhjem and Navrud, 2008) to get a rough idea of the accuracy of the results. It is also advisable to cross-check the scaling-up results with other transfer methods and to check (perhaps informally) whether value unit transformations (e.g. from willingness to pay per household to willingness to pay per hectare) are acceptable in the current context. Whether the accuracy of the final scaled up values is acceptable depends on the purpose and nature of the policy problem as discussed above.

8 Epilogue: a contribution to the economics of ecosystems and biodiversity

The Potsdam Initiative, later TEEB, triggered increased attention on the value of biodiversity to society. Estimating the value of ecosystem services is far from new but the focus has been local or national, rather than international, and addressed partial questions, such as the value of a single species (Chambers and Whitehead, 2003) or an ecosystem (Emerton and Kekulandala, 2003). Exceptionally global values have been addressed but restricted to a single species (Kontoleon and Swanson, 2002). Given the current state of knowledge, the notion of putting a value on conserving all ecosystems at globally seems currently a challenge beyond our means.

Benefit transfer and scaling up have the capacity to broaden our perspective. But precisely what questions can they help answer and how would research on benefit transfer and scaling up help to answer them better? The present report focuses on the policy context. The initiators of TEEB clearly considered support from applied environmental economic analysis useful for making their case: that not halting biodiversity loss would cost society dearly sooner or later. The second phase of TEEB makes that case even more relevant because it aims to inform target groups in a pragmatic sense. TEEB set out to analyse and describe the ecological and economic foundation of the links between biodiversity and ecosystems services, and to inform policy makers at national and local level, consumers and the business community about relevant aspects within their interest and competence (TEEB, 2010).

The core approach of TEEB is not an assessment of biodiversity's total economic value to society. Because biodiversity is essential for human existence, its value is infinite and attempts to estimate it are inherently flawed. Nevertheless, past attempts to put a value on biodiversity as a whole have at least drawn attention to the fact that biodiversity loss entails more than just losing treasured species or 'hotspot' areas of great beauty. Rather, it affects the very foundations of life, as the poor in developing countries already experience daily and as Europeans will find out if the loss is not stopped. Indeed, long before their livelihoods are gravely affected, Europeans will feel the impact of biodiversity loss. As Margot Wallstrøm, then European Commissioner for the Environment, observed, biodiversity matters for ethical, emotional, environmental and economic reasons (Malahilde, 2004).

TEEB sets out to analyse 'the global economic benefit of biological diversity, the costs of the loss of biodiversity and the failure to take protective measures versus the costs of effective conservation' (TEEB, 2010). The context was the EU's policy pledge, agreed during the Gothenburg Summit (2001), to halt the loss of biodiversity in the EU by 2010. The research question then was, what would the global economic loss be when the loss of biodiversity would not be stopped. The COPI study provided the major input to that research question, and specified that question in an outlook analysis. The study took from the OECD Environment Outlook (OECD, 2008) a 'no-new policy' baseline and modelled the changes in land use and biodiversity by 2050 as compared to 2000. The overall economic loss was calculated from the modelled quantitative losses by multiplying these by a per unit price.

How would the results of this report help improving that assessment?

Policymakers need information in four areas. First, how ecosystems services are being underpinned by biodiversity. Second, how changes in biodiversity would affect the quality and resilience of these services. Third, how affected services would change quantitatively. Finally what value these changes have in monetary terms. Economic research takes the first three steps for granted and focuses simply on the fourth area: putting prices on changes in ecosystem services. Such prices are derived from market analysis or using other techniques if no markets exist. Whether using revealed preference or stated preference techniques, deriving non-market prices is a labour-intensive job and becomes impossible when the assessment moves from partial to general, and from local to national, continental or global level. It is in addressing these limitations that value transfer and scaling up may offer benefits.

Ecosystems and the goods and services they deliver will never be identical at the study and the project site, as would ideally be the case for effective benefit transfer. 'Comparability' is a more realistic criterion but less clearly defined. For value transfer to be acceptable from the perspective of economic theory, Loomis and Rosenberger (2006) suggest that three aspects of the project and study sites must be comparable. Specifically, these relate to the ecosystem attributes found there, the market area and the welfare measure used.

The present report suggests ways to reduce errors that arise where ecosystem commodities and market areas are not fully comparable, in particular applying detailed location-specific data in the form of grids.

The assessment in Chapter 6 above of the change in wetland values in the Netherlands between 2000 and 2006 suggests that a meta-analytic value transfer using a large number of characteristics improves the reliability of the assessment. This is because it can control for more site-specific factors than (adjusted) unit transfer and it accounts for changes in the value of ecosystems services, which is important because of their non-linearity. Meta-analytic value transfer takes scarcity into account by controlling for other wetlands in the immediate proximity of the valued wetland.

Several market parameters are relevant for sound benefit transfer. They include the number of people to be marked as potential users of the good under study, the demographic composition, income classification and their specific living situation. As regards recreational values the distance to open space is an important parameter for value assessment. In valuing open spaces, it is not only the magnitude and proximity of cities that matters but also their density because people from spacious neighbourhoods may value open space for recreation less. Demographics count because retired people have more time for recreation. It helps to facilitate benefit transfer when harmonised demographic and economic data are gathered via a nationally accepted census.

This report has highlighted the extensive use of location-specific parameters in the assessment: apart from location, size and scarcity, other attributes that influence the comparability of sites include the proximity of residential areas, the purchasing power of (potential) users or other beneficiaries. It improves the preciseness of valuations, over adjusted unit value transfer that controls for general differences in income between study and policy site, but not for differences in the market for the ecosystems services.

Meta-analytic value transfer may be preferable in some cases, particularly where the study site is geographically larger. The outcome of the transfer is specific for the location-specific variables of the sites, but 'average' for all the other variables captured by the function. This means that the method is more suitable for valuing a larger number of ecosystem sites because differences at individual sites are 'averaged out'.

This report notes that although meta-analytic transfer may be very helpful in scaling up exercises, it can never be the default method. Wherever possible, primary study results should be used, in particular where the study site is located within the boundaries of the larger-scale assessment. It would not be effective to ignore for example the valuation studies of the Wadden Sea (see Chapter 6) in an overall assessment of the value of changes in the Dutch wetlands by meta-analytic transfer, provided double-counting is avoided. That means a bottom-up inventory of available study results should be the core of the data gathering in any larger-scale value assessment.

The overview for wetlands in Europe (see Map 5.1) shows a large incongruence between the location of primary studies and the spread of wetlands in Europe. Primary data are largely lacking for Finland, Ireland and Sweden, countries with vast expanses of wetlands, as well as for the Baltic States and other countries in central Europe. The map also suggests that assessments of ecosystem service values in parts of the Netherlands, the United Kingdom and the Venice region would benefit from primary research.

The meta-analytic value transfer that would probably underpin most large-scale assessments could be refined further. One improvement would be the inclusion of demographic patterns.

Improved large-scale assessment would impose heavy data needs. Available primary studies should be checked for their quality and geographical applicability. Grids of location-specific parameters (land use, demographic, income data) should be made available and adapted to the need of the analysis. The resources required for such a task may be large but would not be as large as those needed for complete primary research.

Ultimately, when primary data are too limited for a scaling up exercise — judged against criteria to be further developed — any value transfer method may lead to unacceptable transfer errors. Primary research would then be due for a reliable outcome.

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2010 – 40 pp. – 21 x 29.7 cm

ISBN 978-92-9213-098-5
ISSN EEA Report series: 1725-9177
DOI 10.2800/41295

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