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From the SPICOSA DOW:

Objective of WP 2.3:

This user guide should be a practical support document for implementing economic assessment in a SAF.

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Executive Summary

This User Guide for Implementing Economic Assessment aims to provide practical guidance to SSAs concerning elements of economic valuation and assessments which are likely to be common to the majority of SSAs.

This report begins with two chapters that are concerned with the theory and practical implementation of benefits transfer (BT). BT is a technique for valuing ecosystem goods that employs results from previously existing studies and transfers them to a similar policy context.

Chapter 2 describes the BT procedure in detail and the various approaches that can be taken. It also includes two examples of the use of BT which particularly relate to the issue of flood control. Chapter 3 is a summary of a study quality assessment procedure for BT that builds upon the more theoretical discussion of Chapter 2 by offering very practical guidelines.

Chapters 4 and 5 deal with issues related to the evaluation of the economic impact of tourism. Chapter 4 is a practical overview of how the economic impacts of tourism might be measured and tourism demand forecasted. Chapter 5 is a very practical description of how multipliers derived from input-output analysis can be used to evaluate the economic impact of tourism (and any other economic sector of interest). It includes an example of the use of this methodology within the Clyde SSA.

Finally Chapter 6 describes how discrete choice models can be used to model behavior and includes two examples of their use within the Himmerfjärden SSA.

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Abbreviations

BT	Benefits Transfer
CBA	cost-benefit analysis
CEA	cost-effectiveness analysis
CGE	Computable General Equilibrium
CM	Choice Experiment
CV	Contingent Valuation
DC	Dichotomous Choice
DOW	Description Of Work
DPSIR	Drivers-Pressures-State-Impact-Response
EA	Environment Agency
EPA	Environmental Protection Agency
GA	green accounting
GDP	gross domestic product
geGDP	greened economy GDP
HP	Hedonic Pricing
I/O	input output analysis
ICZM	Integrated Coastal Zone Management
MCA	multi-criteria analysis
MEA	Millennium Ecosystem Assessment
NAMEA	National Accounts Matrix including Environmental Accounts
NDP	Net Domestic Product
NNI	Net National Income
NPV	net present value
QAI	Quality Assessment Instrument
QOV	Quasi-option value
RUM	Random Utility Model
SAF	System Approach Framework
SAM	Social Accounting Matrix
SCA	supply chain analysis
SEEA	System of Integrated Environmental and Economic Accounting
SNA	System of National Accounts
SSA	Study Site Applications
TCM	Travel Cost Method
TEV	total economic value
TSA	Tourism Satellite Account
WTA	Willingness to Accept compensation
WTP	Willingness to Pay

1 Introduction

1.1 What this document is about

This document has been written by various members of the SPICOSA WP2 team who are responsible for providing the methodology and tools for economic assessment within the project as well as guidelines for inclusion of economic assessment within the System Approach Framework (SAF). Specifically this document represents deliverable D2.3, *User Guide for Implementing Economic Assessment*, as described in the SPICOSA Document of Work (DOW).

The report builds upon the framework for economic analysis that has been described in D2.1 and D2.2 by providing practical guidance concerning methodologies and techniques of economic assessment and through the use of relevant examples.

This report begins with two chapters that are concerned with the theory and practical implementation of benefits transfer (BT). BT is a technique for valuing ecosystem goods that employs results from previously existing studies and transfers them to a similar policy context. We anticipate that very few SPICOSA SSAs will have the resources available to them to undertake new valuation studies for the ecosystem goods they are interested in and so BT may be the only valuation method open to them.

Chapter 2 consists of an article by Roy Brouwer from the Institute for Environmental Studies at Vrije Universiteit, Amsterdam. This details the BT procedure in detail and the various approaches that can be taken. It also includes two examples of the use of BT which particularly relate to the issue of flood control.

Chapter 3 is by Tore Söderqvist of Enveco. This is a summary of a study quality assessment procedure for BT that he developed with other colleagues and which is further detailed in a report produced for the Swedish EPA (this is downloadable from their website as well as the SPICOSA FTP site).

Chapters 4 and 5 deal with issues related to the evaluation of the economic impact of tourism. Chapter 4 is a practical overview of how the economic impacts of tourism might be measured and tourism demand forecasted. Chapter 5 is by Johanna D'Hernoncourt from the Centre for Economic and Social Studies on the Environment at the Université Libre de Bruxelles. This is a very practical description of how multipliers derived from input-output analysis can be used to evaluate the economic impact of tourism (or any other economic sector of interest). It includes an example of the use of this methodology within the Clyde SSA.

Finally Chapter 6 is an article by the team from ENVECO which describes how discrete choice models can be used to model behavior and includes two examples of their use within the Himmerfjärden SSA.

2 Transferability of the Environmental Benefits of Alternative Flood Control

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2.1 Environmental benefits transfer

Environmental benefits transfer is a technique in which the results of previous environmental valuation studies are applied to new policy or decision-making contexts. In the literature, benefits transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the monetary value of similar benefits.

In the field of environmental valuation, benefits transfer has been applied extensively in various contexts, ranging from water quality management (e.g. Luken et al., 1992) and associated health risks (e.g. Kask and Shogren, 1994) to waste (e.g. Brisson and Pearce, 1995) and forest management (e.g. Bateman et al., 1995). Costanza et al. (1997) have extrapolated the monetary values of existing valuation studies to the flow of global ecosystem services and natural capital, and have thereby raised a number of questions as well as heavy criticism about the validity and reliability of benefits transfer.

A number of criteria have been identified in the literature for benefits transfer to result in reliable estimates (e.g. Desvousges et al., 1992; Loomis et al., 1995). These are summarised in Brouwer (2000):

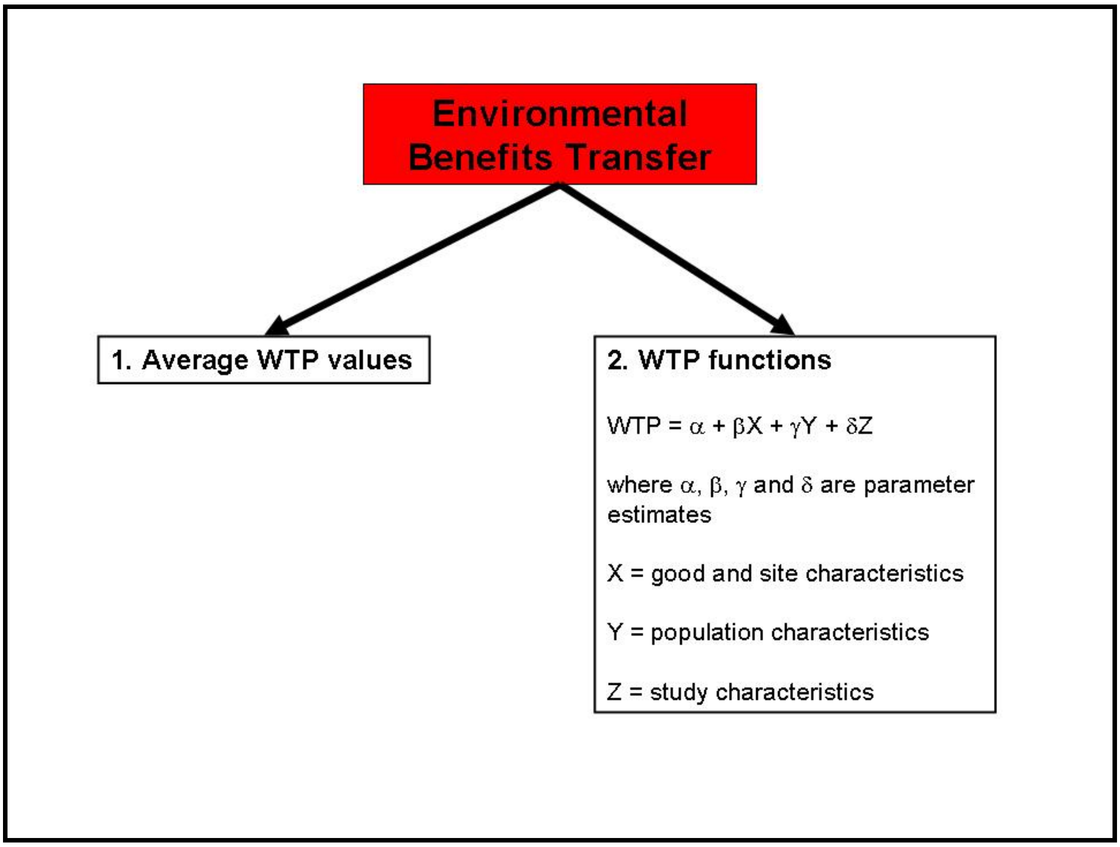
- sufficient good quality data;
- similar populations of beneficiaries;
- similar environmental goods and services;
- similar sites where these goods and services are found;
- similar market constructs;
- similar market size (number of beneficiaries);
- similar number and quality of substitute sites where the environmental goods and services are found.

Study quality is an important criterion, which can be assessed in a number of ways. Above all, one can look at the internal validity of the study results, i.e. the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies. Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated willingness to pay (WTP) function. The reporting of the estimation of the WTP function should also include an extensive reporting of statistical techniques used, definition of variables and manipulation of data.

The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making.

In practice, several approaches to benefits transfer can be distinguished, which differ in the degree of complexity, the data requirements and the reliability of the results. In principle, these approaches are all related to the use of either average WTP values or WTP functions (Box 2-1). The first approach is most frequently applied, as it requires relatively little data or expertise, and is not very time consuming.

Box 2-1: Main approaches to benefit transfer



A first approach is where the unadjusted mean WTP point value is used from another study to predict the economic value of the benefits involved at the policy site. Ideally, this study focuses on the same environmental goods or services, but was carried out at a different location or at the same location at a different point in time.

A second approach is to use and average the unadjusted mean WTP estimates from more than one study, if available, instead of using the result from one study only. These are the two most frequently applied approaches to benefits transfer in practice. They are relatively data extensive and not very time consuming. However, although a quick and cheap alternative, especially compared to original valuation research, the results may be unreliable if circumstances and conditions in the new decision-making context in which they are used are very different from the ones prevailing in the original research.

A third approach is to use one or more mean WTP values adjusted for one or more factors which are, often based on expert judgement, expected to influence the value estimates at the policy site. For instance, mean WTP is sometimes adjusted for differences in income levels at the study and policy site, based on existing information about the income elasticity of WTP for the good or service in question, usually taken from the estimated WTP function in the original study.

A fourth approach is to use the entire WTP function from an original study to predict mean WTP at the policy site. Whereas the three previous approaches are referred to in the literature as ‘unit value’ or ‘point estimate’ transfers, this fourth approach is usually called ‘function transfer’. The estimated coefficients in the WTP function are multiplied by the average values of the explanatory factors in the new policy context to predict an adjusted average WTP value. It has been argued that the transfer of values based on estimated functions is more robust than the transfer of unadjusted average unit values, since effectively more information can be transferred (Pearce et al., 1994). However, this approach is usually more data intensive than the first three as information about all the relevant factors have to be ready available or collected.

A fifth approach is to use a WTP function, which has been estimated based on the results of various similar valuation studies. The difference between this approach and the fourth approach is that the WTP function is in this case estimated on the basis of either the summary statistics of more than one study or the individual data from these studies. In the literature, this approach is usually referred to as meta-analysis. Formally, meta-analysis is defined as the statistical analysis and evaluation of the results and findings of empirical studies (e.g. Wolf, 1986).

Finally a sixth approach can be identified. That is the use of a value function - either one which was estimated in a single previous study (fourth approach) or one which was estimated based on multiple previous studies (fifth approach) - in which the coefficient estimates are adjusted when transferring the estimated value function to a new policy context based on prior knowledge. This approach corresponds to a more Bayesian oriented approach to benefits transfer (e.g. León et al., 2002).

The fourth and fifth function approaches assume that the estimated coefficients remain constant, through time, across groups of people and across locations. However, based on previous knowledge and expert judgement, for instance from previous research at similar study sites or previous research at the new policy site, one may find a reason to adjust coefficient estimates. For example, available information about increases in income level in an area and available information about previously estimated income elasticities of WTP at different income levels, the coefficient estimate in the value function can be modified to better fit the new situation. This approach is expected to become especially relevant when functions are used in benefits transfer exercises, which were estimated a long time ago. Obviously, preferences reflected in stated WTP change as a result of changing circumstances. The fifth and sixth approach can be referred to as an 'adjusted function' approach, because in both cases a new WTP function is used, either based on the adjusted original function or a re-estimated function in a meta-analysis of multiple studies.

Thus, while benefit transfer provides a quick and cheap alternative to original valuation research, some conditions must be met if it should provide reliable results. Above all, the local circumstances and conditions in the new decision-making context need to be close enough to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explaining variables into the transfer, however this also increases the data requirements and the complexity of the analysis. Also, the possibilities of conducting a sound and reliable benefits transfer hinge on the number, quality and diversity of valuation studies available – the larger, the better and the more diverse the existing set of studies is, the more likely will there be a primary study that is "close enough" to the policy site for results to be transferable.

2.2 Uncertainty and transfer errors

The extent to which non-market economic valuation methods are subject to uncertainties and produce estimation errors has not been subject to systematic analysis. In general, a distinction is made in the economic valuation literature between validity and reliability. Validity refers to the question to what extent a method measures what it is intended to measure. This is often called the 'true' economic value of the environmental goods or services involved. Since this true economic value is unknown (the reason why it is being measured through different valuation methods), the validity of economic valuation research is tested in practice by looking at the consistency of research findings compared to the theoretical starting points.[1] Reliability concerns the replicability of findings, for example with respect to the extent to which the method is able to produce the same outcomes at different

[1] In the CV literature a distinction is made between four different validity concepts (e.g. Mitchell and Carson, 1989): content validity, criterion validity, convergent validity and construct validity. It is mainly the last two validity concepts, which have been tested most in the existing literature. A number of studies have compared, for instance, the outcomes of contingent valuation studies with those from travel cost or hedonic pricing studies or other valuation studies (e.g. Smith et al., 1986; Carson et al., 1996) or the outcomes of different WTP elicitation formats in CV such as open ended or dichotomous choice WTP questions (e.g. Desvousges et al., 1983; Bateman et al., 1995).

sites across different groups of people at different points in time. Reliability is usually associated with the degree to which variability in contingent valuation (CV) responses can be attributed to random error.

According to Bateman and Turner (1993), reliability is related to two potential sources of variance: variance introduced by the sample and variance introduced by the method. The usual solution to the former is to use large samples. The general approach in the literature for examining the latter has been to assess the consistency of CV estimates over time in so-called 'test-retest' studies (e.g. Loomis, 1989; McConnell et al., 1998). To date test-retest studies have only considered relatively short periods, ranging from two weeks (Kealy et al., 1988 and 1990) to two years (Carson et al., 1997). These have supported the replicability of findings and stability of values across such modest periods.[2] In a recent test-retest study covering a time period which is more than double that considered in previous test-retest analyses (Brouwer and Bateman, 2005), average WTP values and WTP functions appear to be significantly different across this longer time period for a number of reasons, including those expected from standard economic theory (changes in preferences and incomes).

Although benefits transfer is used extensively in practice, very little published evidence exists about its validity and reliability. Table 2-1 gives an overview of water related studies, which tested the reliability of the transfer of WTP values. Although not complete, Table 2-1 shows that most studies tested the reliability of transferring contingent valuation results. Three studies investigate the transferability of travel cost studies. The estimated benefits in these studies are related to different types of water use, such as recreational fishing, boating or other recreational water use (also the study by Bergland et al. (1995) and Parsons and Kealy (1994) look at water quality improvements for recreational use). The last column presents the range of transfer errors found in these studies, i.e. the absolute error when using the estimated economic value of a specific water use or water quality deterioration from another study in a new policy context. So, a transfer error of 50% means that the value from the previous study used in the new policy context is 50% higher or lower than the 'true' value in the new policy context. A range of transfer errors is presented as the reliability of benefits transfer was tested for at least two sites (transferring a WTP value from say site A to site B and the other way around) and for both WTP values and WTP value functions (see Brouwer (2000) for more details).

From Table 2-1, it is difficult to say how large the errors can be expected to be on average when using existing economic value estimates in new decision-making contexts. In some cases they can be very low, in other cases they can be as high as almost five times the value, which would have been found if original valuation research was carried out. No distinct differences can be found based on Table 2-1 when comparing transfer errors for contingent valuation and travel cost studies.

[2] An overview of studies investigating the reliability of CV estimates is found in McConnell et al. (1998).

Table 2-1: Errors found in water related economic valuation studies testing benefits transfer

Study	*Valuation method*	*Estimated benefits*	*Transfer errors (%)*
Loomis (1992)	TC	sport fishing benefits	5 – 40
Parsons and Kealy (1994)	TC	water quality improvements	1 – 75
Loomis et al. (1995)	TC	water based recreation	1 – 475
Bergland et al. (1995)	CV	water quality improvements	18 – 45
Downing and Ozuna (1996)	CV	saltwater fishing benefits	1 – 34
Kirchhoff et al. (1997)	CV	white water rafting benefits	6 – 228
Brouwer and Bateman (2005)	CV	flood control benefits	4 – 51

Source: Adapted from Brouwer (2000).
Notes: TC= Travel Cost, CV = Contingent Valuation

Another illustration of the accuracy underlying the use of existing economic estimates as proxies for environmental values is presented in Table 2-2.

Table 2-2: Break-down of average economic values found in the literature for wetlands in temperate climate zones in US$ per household per year (price level 1995)

	Mean WTP	*Standard error*	*Min WTP*	*Max WTP*
Wetland type				
Saltwater	84.3	40.8	28.5	205.5
Freshwater	88.4	9.2	1.5	400.5
Wetland function				
Flood water retention	138.9	36.6	36.0	265.5
Water recharge	32.3	10.2	4.5	88.5
Pollutant retention	78.8	8.9	13.5	261.0
Wildlife habitat	114.2	19.2	1.5	516.0
Wetland value				
Use value	102.2	12.6	13.5	516.0
Non-use value	53.3	7.2	18.0	117.0
Use and non-use	95.7	19.4	1.5	400.5
Continent				
North America	106.2	11.7	4.5	516.0
Europe	49.2	12.6	1.5	265.5

Source: Adapted from Brouwer et al. (1999).

Table 2-2 presents an overview of the results of a meta-analysis of 30 CV studies of wetlands in temperate climate zones. The CV studies focus on different issues related to wetland conservation and were carried out at different points in time (in the 1980s and 1990s) in different places (different countries in Europe and North America). A statistical meta-analysis of the findings of the different CV studies produced the summary statistics shown in Table 2-2.

The summary statistics (average WTP values) show a high degree of variability (measured through the minimum (Min) and maximum (Max) average WTP values found in individual studies). Standard errors, measures of the accuracy of the estimated average values, range between 10 and 50 percent of the summary statistic's average value (i.e. variation coefficient). The 95 percent confidence interval around these estimates is almost two times higher. For instance, the 95 percent confidence interval around the average economic value of freshwater wetlands is US$ 70.4 – 106.4, whereas the 95 percent confidence interval around the average economic value of saltwater wetlands is US$ 4.3 – 164.3. Together with the hydrological function water recharge, floodwater retention has the highest variation coefficient. The variation coefficient related to the economic value of the ecological function wildlife habitat provision is about half the size of that. However, the range of values found in the existing literature is highest for this latter ecological function, varying between one and five hundred US dollars per household per year.

The errors reported in Table 2-1 have to be considered in the light of the purpose the user wishes to use previous valuation results for. In some cases the user may find a transfer error of 50 percent too high, in other cases such an error may be acceptable. The extent to which the transfer errors reported in Table 2-1 are considered a problem depends upon the acceptability of these errors by the user (policy or decision maker) of the results. User acceptability of these errors will depend upon subjective judgement by the user self, but also on the purpose and nature of the cost-benefit evaluation and the phase of the policy or decision-making cycle in which the evaluation is carried out. The reliability (and corresponding errors) of pre-feasibility studies carried out in an early stage of policy formulation to aid policy development is usually much lower (and errors larger) than the reliability of detailed cost-benefit studies which are looking at the practical implementation of concrete policy measures on the ground. In general, the further the policy or decision-making process has moved forward towards practical implementation, the higher the reliability of the evaluations based on increasing quantity and quality of information. Large errors and low reliability as a result of unresolved uncertainties and lack of information will become less and less acceptable the closer the project moves towards the practical implementation of policy measures on the ground.

2.3 Trying to explain non-transferability and large transfer errors

A number of reasons have been suggested in the literature why the test results found so far are ambiguous (Brouwer, 2000). First, contrary to many of the market based costs and benefits included in cost-benefit analysis (CBA), environmental values are not always well defined, especially in situations where the complexity of the environmental issue extends the complexity of the valuation process beyond reasonable expert and/or public comprehension.

This undermines their political and legal acceptability in CBA, especially in those cases where they seriously inflate total benefits (costs) for green (economic development) programmes. In the case of travel costs and hedonic pricing studies, it is usually fairly clear what is measured: a use value revealed through the amount of money people actually paid to enjoy an environmental good or service. On the other hand, in the case of CV expressed WTP values may have a variety of meanings, related to (potential) use and non-use. In fact, they may be so diverse that attempts to aggregate them across individuals to produce a total economic value ultimately obscure what exactly is measured. The problem of correctly interpreting findings on the basis of underlying motivations has sometimes been referred to as a ‘technical’ survey problem of proper definition of the good being valued. However, it may also reflect people’s inability to express much more than a general moral commitment to help financing environmental programmes (Vadnjal and O’Connor, 1994).

A wide range of values produced by a black box undermines the argument put forward to include those values, especially non-use values, which reflect some kind of overall moral commitment to environmental causes and which are expected to stay more or less the same across social groups and environmental domains. If more or less constant, these values would be easily transferable without a need to look at motivations underlying such WTP values. However, values often do differ substantially in practice from case to case.

Secondly, as a result of unclear definition, there is a real risk of double counting when aggregating these values across different stakeholder groups.

Thirdly, instead of solving the problem of aggregation (i.e. the number of stakeholders and the values they hold to be included in the analysis), the inclusion of especially non-use values only seems to aggravate the problem. They show that also non-users may attach a value to the environmental goods and services involved, but without identifying the boundaries of this specific ‘market segment’. On the other hand, CV values elicited in a very specific local context based on a sample of local residents or visitors may also reflect more than simply current and future use values. The historical-cultural context in which these values have come about may be a significant determinant of the elicited WTP values. Also in those cases where stated values seem to reflect upon well-defined local issues, it is important to carefully investigate the broader applicability of these values which may be embedded in specific local conditions when aiming to transpose these values across sites.

Finally, it is perhaps also important to point out that especially CV results reflect a one-time snapshot of people’s preferences. Evidence furthermore suggests that CV surveys evoke constructed rather than well-articulated preferences, especially in situations where people are unfamiliar with a specific environmental issue or are asked for a maximum WTP for public goods. Preference and value formation on the basis of the information supplied is not specific to CV, but a more general phenomenon in communication consistent with findings in socio-psychological research of decision-making (Schkade and Payne, 1994). However, the question is how stable constructed preferences and subsequently people's stated WTP in a say 15 to 30 minute interview remain through time and subsequently how legitimate it therefore is to put them together and make them comparable with other value statements at different points in time in a discounted CBA. One could argue the same for market based costs and benefits reflecting existing market prices. Also these costs and benefits are assumed to stay the same through time. However, for these prices often time series are available, which can be analysed and extrapolated.

Finally, the explanatory power of most benefit functions usually does not exceed 30 percent. Although R-squared statistics have to be interpreted with the necessary care in view of the nature of the panel data collected in economic valuation research, most of the variability in stated WTP amounts remains unexplained. Therefore, perhaps not surprisingly, a generally applicable model has not yet been found. The quantity and quality of control included in most models is very limited in terms of the way general site and population characteristics are specified statistically, for instance as dummy variables which merely indicate whether or not a site is accessible to the public or someone earns a specific amount of income. This simple specification of explanatory factors is in sharp contrast with the complex continuous response variable, which is expected to reflect the strength of people's preferences for specified changes in provision levels of environmental goods and services.

Furthermore, even if statistically specified adequately, most factors included in these models do not explain why respondents from the same socio-economic group may hold different beliefs, norms or values and hence possess different attitudes and consequently state for instance different WTP amounts, especially in a CV study. Human behaviour as measured in travel cost studies and hedonic pricing studies and behavioural intentions as measured in CV are liable to several influencing factors, as can be learned from the related socio-psychological literature (e.g. Brown and Slovic, 1988). Attitudes are considered an important key to the understanding of people's preferences in terms of WTP (Fishbein and Ajzen, 1975). However, if accounting for attitude variables provides a valid basis for value transfer, then this is bad news for its practical viability since it suggests the need for data collection of such variables alongside people's socio-economic characteristics at the policy site. The data needed to calculate adjusted average value estimates based on a value function at the policy site has to be easily accessible for value transfer to remain a cost-effective valuation alternative.

2.4 Economic values of environmental benefits related to natural flood control ecosystems

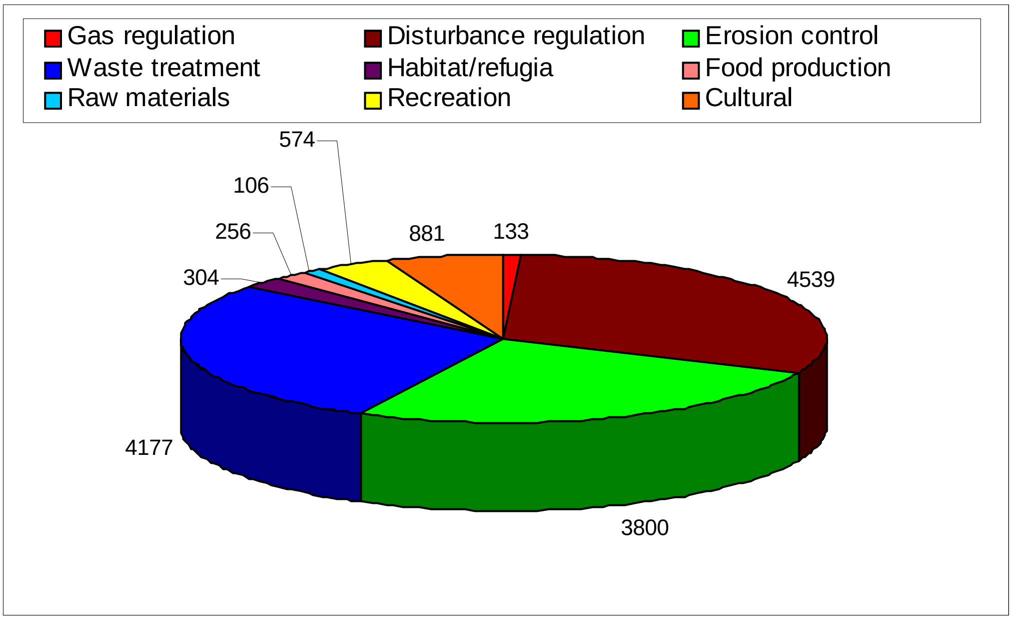
Wetlands and floodplains have long been regarded by society as having very little or even negative value, often being described as wastelands or sources of disease. As a result, they have been drained and converted into other uses, while the essentially open nature of wetland and floodplain systems has made them susceptible to indirect damage from other human activities. This has led to the stock of wetlands, particularly in Europe, being substantially diminished. According to the European Environment Agency (EEA, 1999), wetlands and floodplains continue to be under particular pressure because of the extensive drainage of lowland areas for agriculture, forestry, peat exploitation and urban development, together with the impacts of river system regulation for power generation, water storage and flood control, and the maintenance of navigation channels. Over the past decades, it has become apparent that wetlands and floodplains, far from being valueless, perform a wide array of functions that can be of considerable value to society (Box 2-2).

Over the past years, a number of studies have looked at the economic values associated with the environmental benefits provided by wetlands and riverine floodplains. Meta-analysis techniques of wetland and floodplain ecosystem valuation studies have been carried out by Brouwer et al., 1999, Woodward and Wui (2001) and Brander et al. (2006). The results from these three different studies are presented in Table 2-3. Brouwer et al. (1999) distinguished four main wetland functions, which were valued in economic terms in different empirical studies across Europe and North America:

- flood water retention (flood control)
- surface and groundwater recharge (water quantity)
- nutrient retention and export (water quality)
- wildlife habitat and nursery

Box 2-2: Total economic value of wetland ecosystem services worldwide in 1994 US$ $ha^{-1}yr^{-1}$

In 1997, an attempt was made to put a monetary value on the various goods and services provided by the different world's ecosystems, including wetlands (Costanza *et al.,* 1997). The total economic value of wetlands was estimated at US$ 14,785 per hectare per year (price level 1994). This total economic value consisted of 10 different functions, nine of which are presented in the pie diagram below. The water regulation function (i.e. regulation of hydrological flows such as the provision of water for agricultural or industrial processes or transportation) is not included in the diagram because of its relatively low value (US$15 ha^{-1} yr^{-1}) compared to the other functions. The estimated total economic value of the functions provided by wetlands in this study is the second highest value after marine coastal estuaries, and higher than the total economic value of lakes and rivers, coral reefs or tropical forests.



- Gas regulation: regulation of atmospheric chemical composition (e.g. CO_2/O_2 balance)
- Waste treatment: recovery of nutrients and removal or breakdown of excess nutrients and compounds
- Raw materials: gross primary production extractable as raw materials (e.g. lumber, fuel or fodder)
- Disturbance regulation: capacitance and damping of ecosystem response to environmental fluctuations
- Refugia: habitat for resident and transient populations
- Recreation: providing opportunities for recreational activities (e.g. sport fishing)
- Erosion control: retention of soil within an ecosystem
- Food production: gross primary production extractable as food (e.g. production of fish, crops etc.)
- Cultural: opportunities for non-commercial uses (e.g. aesthetic, artistic, educational, spiritual and/or scientific values of ecosystems)

Table 2-3: Comparison of findings in three different meta-analyses of wetland valuation studies

	Brouwer et al (1999)		Woodward & Wui (2001)		Brander et al (forthcoming)	
	Variables	**Coefficient**	**Variables**	**Coefficient**	**Variables**	**Coefficient**
Dependent variable	Mean WTP (1990 SDRs)/hh/year		1990 USD/acre/year		1995 USD/ha/year	
Constant	Intercept	3.356****	Intercept	7.872**	Intercept	-6.98
Socio-economic					GDP per capita (ln)	1.16*
					Population density	0.47***
Wetland type			Coastal	-0.117	Salt/brackish marsh	-0.31
					Unvegetated sediment	0.22
					Mangrove	-0.56
					Freshwater marsh	-1.46**
					Freshwater wooded wetland	0.86**
Wetland function	Flood control	1.477****	Flood control	0.678	Flood and storm protection	0.14
	Water supply	0.691**	Water supply	0.737	Water supply	-0.95
	Water quality	0.545*	Water quality	-0.452	Water quality	0.63
			Recreational fishing	0.582	Recreational fishing	0.06
			Bird hunting	-1.055**	Recreational hunting	-1.10**
			Amenity	-4.303**	Amenity	0.06
			Habitat and nursery	0.427	Habitat and nursery	-0.03
			Storm protection	0.173	Materials	-0.83**
			Bird watching	1.804**	Fuelwood	-1.24***
			Commercial fishing	1.360	Biodiversity	0.06
Wetland size			Acres (ln)	-0.286**	Hectares (ln)	-0.11**
Continent	North America	1.861****				
					South America	0.23
					Europe	0.84
					Asia	2.01
					Africa	3.51**
					Australasia	1.75*
Other geographic characteristics					Latitude (absolute value)	0.03
					Latitude squared	-0.00
					Urban	1.11**
Valuation method	CVM Open-ended	-0.411***			CVM	1.49**
			Hedonic pricing	5.043**	Hedonic pricing	-0.71
			Net factor income	0.273	Net factor income	0.19
			Replacement cost	2.232**	Replacement cost	0.63
			Travel cost	-0.341	Travel cost	0.01
					Production function	-1.00
					Opportunity cost	-0.03
					Gross revenues	-0.04
Payment vehicle	Income tax	1.880****				
Welfare measure			Producer surplus	-3.140**		
Study quality	Response rate (39-50%)	-2.253****	Published	-0.154		
	Response rate (>50%)	-1.904****	Data	0.000		
			Theory	-1.045		
			Econometrics	-3.186**		
Year of study			Year	0.016		
Other variables					Ramsar proportion	-1.32*
					Marginal value	0.95*
	Pseudo R2	0.365	R2	0.582	Adjusted R2	0.45
	n	92	n	65	n	202

Woodward and Wui (2001) and Brander et al. (2006) distinguish already many more wetland functions, valued through a wider variety of economic valuation methods:

- flood control
- water supply
- water quality
- recreational fishing
- commercial fishing
- recreational hunting
- bird watching
- amenity
- habitat and nursery
- materials
- fuel wood
- biodiversity

Brander et al. (2006) also included tropical wetlands. The units in which the economic values of different wetland ecosystem functions are measured also differ between the three studies. Brouwer et al. (1999) measures the economic values in terms of money units per household per year, whereas Woodward and Wui (2001) and Brander et al. (2006) measure these values in terms of money units per land unit (hectare or acre) per year. Woodward and Wui (2001) conclude that ‘while some general trends are beginning to emerge, the prediction of a wetland’s value based on previous studies remains highly uncertain and the need for site-specific valuation efforts remains large’.

Expressing ecosystem values in money terms (and per hectare) has evoked quite a bit of discussion worldwide, especially after the publication of the paper by Costanza et al. (1997) in Nature, in which an attempt was made to value all global ecosystem services, including those provided by wetlands (see Box 2-1). The discussion about the validity and reliability of economic valuation of ecosystems and the goods and services they provide is on-going, also specifically for wetlands (e.g. Clark et al., 2000).

2.5 Towards a protocol of good practice

In principle, the reliability of benefits transfer can be approached from two main perspectives. The first one, which has been dominating the value transfer literature so far, does not question the environmental values themselves. The monetary values found are taken as valid and reliable outcomes of people’s valuation. The variability found in valuation outcomes is attributed to differences in study design, good and population characteristics and to some extent value types (use and non-use values). Hence, there may be something wrong with, for instance, the value elicitation mechanisms used, but the values themselves remain undisputed.

A second perspective, advocated in Brouwer (2000) as a complementary approach to the first one, is more critical about the estimated values. Even though a valid transfer can be established when the explanatory power of the transfer model is low (Brouwer and Spaninks, 1999), the question is whether users of environmental valuation results are happy with the numbers they are given from a 'black box'. How can environmental values be reliably predicted across sites and people if currently much if not most of the variability of the values in original studies can not be explained? This second perspective is focusing much more on the process of value formation, articulation and elicitation in order to better understand the values themselves.

Based on these premises, a number of steps will be highlighted which are considered important to the practice of environmental benefits transfer and monetary valuation of environmental change in general. If previous study results are questionable in terms of validity and reliability, their use in new policy contexts will only result in more controversy. The steps are summarised in Box 2-3.

Box 2-3: Practical steps towards a protocol for good practice

Step 1: Defining the environmental goods and services to be valued
Step 2: Identifying stakeholders and/or beneficiaries
Step 3: Identifying the various values held by different stakeholder groups and/or beneficiaries
Step 4: Stakeholder involvement in determining the validity of monetary environmental valuation
Step 5: Study selection
Step 6: Accounting for methodological value elicitation effects
Step 7: Stakeholder and/or beneficiaries involvement in value aggregation

Step 1: Defining the environmental goods and services

An essential part of the first step is the identification of the relevant ecological functions which underpin the supplied goods and services and the importance of these functions for sustaining ecosystems and hence human systems. Obviously, this requires scoping of the problem in terms of the geographical and temporal scales involved.

Environmental goods and services provide different kinds of benefits to different kinds of people. In order to keep the analysis transparent and to avoid double counting, the benefits preserved or foregone have to be identified first, for example in terms of direct and indirect extractive and non-extractive benefits. Examples of direct extractive benefits from renewable natural resources are fish and wood, while examples of direct non-extractive benefits are recreational activities in forests, rivers or lakes. Indirect benefits are often found off-site. An example of an indirect extractive benefit from renewable resources is clean drinking water, while an example of an indirect non-extractive benefit is the provision of landscape diversity.

For the purpose of a valid and reliable benefits transfer, the identification of the various economic benefits is not enough. The provision and quality levels of these benefits in the reference and desired target situation is equally important (Fischhoff and Furby, 1988). In practice, reference and target situations in the old and new policy context may differ significantly, seriously limiting the application of previous study results across different policy contexts. Most CV studies lack information about preferences for a variety of reference and target levels, hence the recent increase in popularity of multi-attribute utility based choice models. In the case of CV, no adjustment mechanism is available to account for possible differences. Random utility travel cost models and contingent choice experiments seem to be the only tools available at present which are able to meet this problem.

Step 2: Identifying stakeholders

Different benefits usually accrue to different groups of people. After the various benefits preserved or foregone have been identified, the people who value these benefits for what they are, the beneficiaries, have to be identified. Although this step identifies beneficiaries, not the reasons why these beneficiaries value environmental goods and services (see the next step), they are interdependent. To clarify this, an analogy with market goods and services can be drawn. When estimating the economic value of market goods and services, an important step is to look at their market size in order to determine which prices should be used in the value calculation, for example local market prices or world market prices. In principle, one could argue that the same applies to non-market goods and services.

Step 3: Identifying values held by different stakeholder groups

The same good or service may hold different values to different people. An analogy can be made again with market goods and services: within the market place different market segments may exist where different prices prevail. When identifying the benefits of environmental goods and services, the reasons why these benefits are considered benefits by various stakeholders has to be addressed at the same time. Benefits can only be identified as such if their value is known. Whether or not this value can be monetised is another question (see the next step).

Step 4: Stakeholder involvement in determining the validity of monetary environmental valuation

One of the underexposed areas in monetary and non-monetary environmental valuation is the assessment of the appropriateness of different valuation procedures in different environmental domains based on their underlying axioms and assumptions. Like traditional economic theory, alternative approaches to environmental valuation based on social processes of deliberation may be questioned on their implicit value judgements regarding the legitimacy of the social-political organisation of the process of value elicitation. Instead of making assumptions a priori, research efforts should perhaps focus more on the processes by which actual public attitudes and preferences towards the environment can best be facilitated and fed into environmental or other public policy decision-making.

One way of making sure that the transfer (valuation) exercise generates socially and politically acceptable results is to get the stakeholders involved who are (going to be) affected by environmental change and whose values the researcher and decision-maker(s) are interested in. This stakeholder consultation process provides the researcher with an external validation mechanism of the monetary environmental valuation exercise and helps defining the boundaries of monetary environmental valuation. Stakeholder groups or their representatives can be asked for their most preferred form of public consultation in general and environmental value elicitation in particular before any value elicitation structure is imposed on them. If there is agreement about the monetisation of certain environmental values present in a specific policy context, stakeholder involvement can be very useful in determining what these monetary values should reflect (e.g. in terms of individual WTP). It is then up to the researcher to look into previous studies and see to what extent these values have been estimated in a valid, reliable and, if possible, replicable way.

There usually is increased difficulty in computing monetary economic values from direct extractive to indirect non-extractive benefits. Monetary values for direct extractive benefits (e.g. fish, reed etc.) can often be computed from available market data. In some cases, market data will also be available for indirect extractive benefits (e.g. water consumption off site). In other cases, one can rely upon non-market valuation techniques. Direct non-extractive values (e.g. recreational benefits) are more difficult to calculate since market data will be absent unless one relies upon some complementary relationship between the non-market benefit and for example actual expenditures made to enjoy the good (as in travel cost studies). Finally, indirect non-extractive benefits are usually the most difficult benefit category to estimate in money terms. Market data will not be available and there may exist a whole range of diverse reasons why people value these benefits, which may be difficult to accommodate in money. CV is usually the only way to estimate these benefits.

Step 5: Study selection

After going through steps 1 to 4, appropriate studies have to be selected. If possible or available, a meta-analysis of these studies will provide a useful tool to synthesise previous research findings, for example by identifying different outcomes as a result of different research design formats. Otherwise, a number of criteria have been identified in the literature to select among studies (Desvousges et al., 1992; Loomis et al., 1995). These criteria are generally applicable (see section 2.1). Often the selection of existing studies will be based on a qualitative assessment. Study quality is an important criterion, which can be assessed in a number of ways.

First, one can look at the internal validity of the study results, i.e. the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies. Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated WTP function. The reporting of the estimation of the WTP function should also include an extensive reporting of statistical techniques used, definition of variables and manipulation of data.

Secondly, the appropriateness of monetising environmental values in a specific context through individual WTP, i.e. their external validity, can be assessed by looking at the actual meaning and interpretability of the values found. Contrary to TC and hedonic pricing (HP), CV allows assessment of the external validity of stated WTP values through the social survey format itself: i.e. via response rates, protest bids and reasons why respondents are willing and able to state a specific payment.

Response rates are often ill-defined in the reporting of CV results. A high non-response, either to the entire survey instrument or the valuation question, raises concern regarding the study's representativeness, and questions the validity of the survey design employed and the extent to which the valuation scenario in the questionnaire was comprehensible and credible (Arrow et al., 1993).

Criteria to determine whether or not a respondent is a legitimate zero bidder to a WTP question or a protest bidder are often arbitrary. A lot of studies do not report these criteria at all. No guidelines exist as to how much protest responses invalidate a survey. It is common practice to exclude them from further analysis, classifying them as 'non-usable response' without providing detailed information why respondents protested. Protest responses reveal much more useful information than they have been given credit for in CV research. They can be used as an indicator of the acceptability of the use of the monetary environmental values by different stakeholder groups.

Asking respondents for the reasons why they protest against the WTP question or why they are willing and able to state a specific payment is considered of paramount importance to assess the appropriateness of the survey and the actual meaning of their replies. Understanding the meaning of answers, especially to the valuation questions, is a prerequisite to define the appropriate context in which the survey results can be used and how they should be interpreted. Therefore, besides thorough pre-testing of survey formats, it is recommended that post-survey debriefings of interviewers and respondents are used, individually or in a group, to discuss the actual meaning of the answers given in the questionnaire.

Step 6: Accounting for methodological value elicitation effects

Different research designs in environmental valuation methods such as TC, HP and CV have resulted in different results. In TC and HP models, most of the differences seem to originate from the specific model used, the statistical estimation method, the inclusion or exclusion of specific explanatory variables, the definition of these variables and data quality. It is difficult to recommend adjustment mechanisms for these differences in research design. For instance, which statistical model specification is expected to provide the most robust results? RUMs provide certain advantages over the traditional zonal TC models, but at the same time there is an increase in complexity with respect to the statistical models used, the assumptions underlying the computational heuristics of these models and their data requirements. This also applies to most contingent choice experiments and CV studies using iterative bidding formats.

In CV different survey elements have been shown to result in different WTP values. A number of research design effects have been investigated in the past, of which payment mode, elicitation format, the level of information, sensitivity to scope and/or embedding effects are probably the most important ones. As in TC and HP models, it is often hard to tell how CV findings should be modified based on the specific research design used. In accordance with best practice recommendations, generally a conservative approach seems to be preferable (Arrow et al., 1993).

Step 7: Stakeholder involvement in value aggregation

After one or more studies have been selected and values are found which reflect the values policy or decision-makers are looking for under the specific circumstances, these values can be adjusted, if necessary and secondary data at the policy site are available, for differences in site and population characteristics using the estimated WTP function or average WTP value. These modified values can then be discussed again with (representatives of) different stakeholder groups to which they relate before they are extrapolated over the relevant population which is (going to be) affected by the environmental change. Also this should be discussed with the stakeholder groups involved. Finally, the economic aggregate is included in a CBA together with other economic costs and benefits, which can then play its part in the facilitation of the overall, real world, multi-criteria decision-making process.

2.6 Examples

2.6.1 Technical approach

This first example looks at the transferability of visitor valuation of the recreational and amenity benefits provided by the Broads National Park, one of the most extensive freshwater wetlands in the UK. The example is based on Brouwer and Bateman (2005). More specifically, this example illustrates the stability of WTP values and WTP functions over an extensive period of time. The example considers a time period between surveys which is more than double that considered in previous test-retest analyses. Whereas such previous studies have reported stable values over relatively short time periods, the example presented here finds a statistically significant decrease in real WTP over this more extended time period. The issue of temporal stability over extended periods is one of more than academic interest. CBAs frequently employ values estimated some considerable time prior to those analyses. Temporal stability is therefore implicitly assumed rather than explicitly tested. Yet there is no reason to suppose that values for non-market goods should remain constant over extended periods.

Temporal stability is addressed through the application of two matching surveys, concerning the same case study area (the Norfolk Broads in the UK), focusing on the same environmental good and valuation scenarios (flood protection and conservation of freshwater wetland habitat and associated recreational amenities), using the same payment vehicle (coercive taxation), the same sampling frame (random in-person interviews) applied to the same sample population (visitors to the area), but sampling at different points in time, namely in the summers of 1991 and 1996.

The Norfolk Broads are a large freshwater wetland area located in East Anglia, UK. The area consists of a system of shallow lakes, marshes and fens, linked by low-lying rivers. The site is of national and international wildlife importance, being a designated Environmentally Sensitive Area (ESA) and containing twenty-four Sites of Special Scientific Interest (SSSI), including two sites notified under the international RAMSAR convention. The area is also a major focus for recreation, attracting more than one million visitors a year, of which 200,000 spend their holidays on boats hired for a week or longer (Broads Authority, 1997).

The character of the low-lying landscape of the Broads depends upon 210 km of reinforced river embankments for protection from saline tidal waters. However, at the time of the surveys these flood defences were increasingly at risk from failure, because of their age, erosion from boat wash and the sinking of the surrounding marshlands. Thus, the standard of flood protection afforded by these man-made defences was decreasing over time. If flood defences were breached, the ensuing saline inundation would fundamentally and enduringly alter the nature of the area both in terms of its habitat capabilities and in respect of the recreational opportunities currently afforded.

In 1991 the National Rivers Authority (NRA), later named the Environment Agency (EA), initiated a wide-ranging ‘Flood Alleviation Study’ to develop a cost-effective strategy to alleviate flooding in the Norfolk Broads for the next 50 years (Bateman et al., 1992). The flood alleviation study consisted of five main components: hydraulic modelling; engineering; benefit-cost assessment; environmental assessment; and public consultations. The item of most relevance here is the benefit-cost assessment, which compared benefits of undertaking a scheme to provide a particular standard of flood protection to the corresponding costs. Although market benefits of flood alleviation have been considered in terms of agriculture, industry, living conditions and infrastructure (Turner and Brooke, 1988), the value of the non-market benefits from the area was uncertain.

As part of the benefit-cost assessment, a large CV study was mandated in 1991 (Bateman et al., 1994; 1995) and a follow-up carried out in 1996 (Powe, 1999; Powe and Bateman 2003; 2004), in order to assess user valuations of conserving the area in its current state. The studies aimed, among other things, to provide a valid and reliable monetary estimate of the current recreational and amenity benefits enjoyed by visitors to the Broads. Findings were used to inform a CBA of various flood defence options (Brouwer et al., 2001). The cost-benefit ratio found ranged between 0.98 and 1.94 (National

Rivers Authority, 1992). The results, including the findings from the 1991 CV study, were submitted to the relevant Ministry of Agriculture, Food and Fisheries as part of an application of central government funding support for the proposed flood alleviation strategy. Following lengthy consideration of this application, the EA received conditional approval for a programme for bank strengthening and erosion protection in 1997 (Environment Agency, 1997). The actual scheme was taken forward in 2000 on the basis of a long-term private-public partnership scheme between the EA and relevant government support ministries and a private engineering firm consortium.
Temporal reliability of the dichotomous choice CV models estimated in this study was tested by examining the statistical equality of unadjusted average WTP values (hypothesis 1) and the DC WTP functions (hypothesis 2). An iterative approach was developed in order to see how much control is needed to produce transferable models of WTP. These models are generated by progressively blending theoretically expected determinants of WTP with additional ad-hoc variables, which may be more transitory in their effect. This approach involves a gradual expansion in the number of explanatory variables added to a model of WTP. At each addition of a variable temporal transferability is assessed by applying the model to both the 1991 and 1996 data and undertaking various tests. This progressive expansion approach should in principle allow the identification of the optimal level of control for transferability. This approach is compared to that obtained by estimating a statistical best-fit model for a given dataset and transferring this to the other survey period and vice-versa.

For each model transferability is assessed both forward in time (from 1991 to 1996) and back (from 1996 to 1991) using statistical tests for coefficient stability as per Brouwer and Spaninks (1999). A further test of the transferability of each specification is obtained by pooling the data and assessing transferability through application of the Likelihood Ratio (LR) test as per Downing and Ozuna (1996) and Carson et al. (1997). For this latter test data from the two surveys are pooled and a dummy variable included to represent the year in which the study was undertaken. If study year has a significant impact on respondent WTP, this implies that the study results are not transferable. The pooled regression results are the same as the outcomes of the LR test.

Mean WTP values based on parametric and non-parametric estimation approaches are presented in Table 2-4. In order to be able to compare the 1991 and 1996 WTP values, the 1996 values are corrected for intervening differences in purchasing power.

Table 2-4: Mean real WTP values from the 1991 and 1996 surveys (£ p.a. in 1991 prices) obtained from the parametric logistic model and (lower bound) non-parametric Turnbull model

	Parametric Linear-Logistic		***Non-parametric Turnbull***	
	1991	**1996**	**1991**	**1996**
Mean WTP (£)	248.1	215.8	54.2	37.8
Standard error	23.3	29.3	2.9	2.4
95% CI {1996 – 1991}	{-34.3 ; -30.3}		{-16.6 ; -16.2}	
Min-max values	-∞ - +∞	-∞ - +∞	0-200	0-200
N	1747	1108	1747	1108

The results from the linear-logistic and Turnbull models suggest that visitor valuation of the recreational and amenity benefits provided by the Broads has decreased across the period between the two surveys. In constant prices, mean WTP calculated from the linear-logistic model is 13 percent lower in 1996 than in 1991, and 30 percent in the case of the Turnbull model. The observed difference in income levels between the 1991 and 1996 visitors is one possible explanation for this decrease.

Although the Turnbull model is known to provide a lower bound for mean WTP, the large difference between the Turnbull and linear-logistic model is striking. The parametric estimates are about five times higher than the non-parametric estimates. No big differences exist in terms of the accuracy of the estimates. In relative terms the standard errors of the linear-logistic estimates are only slightly higher than the standard errors of the Turnbull estimates. The differences in mean WTP are statistically significant as can be seen from the 95 percent confidence interval (CI) constructed around their difference based on the standardised normal variable (z). The estimated differences indicate that the real value of the recreational amenities in the Broads have decreased by 3 to 6 percent per annum over the study period. This significant decrease in real WTP is in contrast to the non-significant changes noted over shorter periods and may well be a consequence of the longer interval under consideration in this example.

Results from our various analyses of model transferability are shown in Table 2-5. From Table 2-1 it can be observed that, using the LR test, all models appear transferable. However, adopting the Wald test (which is more stringent) yields a more mixed result, but one from which a clear pattern emerges. Focusing upon these latter tests, both models relying solely upon variables suggested by economic theory (models using the Bid variable alone or those supplementing this with the household Income variable) are transferable. However, when such models are extended through the addition of more ad-hoc variables, not derived from theory, transferability becomes sporadic. Here, those models using the binary Local variable (identifying those respondents who live near to the study site) do transfer, whereas those substituting in the continuous Distance variable (the number of miles travelled to reach the site) fail Wald tests of transferability, questioning the usefulness of more sophisticated distance-decay relationships in models of WTP for transfer purposes. Statistical best-fit models (see the Annex) also fail Wald transferability tests. This reflects the differing determinants, which enter each of these models.

Hence, while previous studies considering shorter periods have shown no significant difference in real WTP values, the analysis presented here reveals a significant difference across a longer period of time. Tests of model transferability indicate that simple models, based solely upon variables derived from economic theory, are transferable across this period. This suggests that underlying relationships for such key determinants are stable even across this longer period. However, expanding models by including theoretically unanticipated factors brings ad-hoc and possibly transitory factors into the models, which consequently prove non-transferable.

Table 2-5: Transfer test results from the dichotomous choice CV models

		Model specification							
Transfer	***Test***	***Bid***	***Bid Income***	***Bid Income Distance***	***Bid Income Local***	***Bid Income Distance Scenery***	***Bid Income Local Scenery***	***Best fit 1991***	***Best fit 1996***
Transfer of the estimated 1991 models to 1996	Wald	0.93	3.71	9.70	3.51	13.20	5.88	20.50	15.03
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
	LR	0.58	2.19	6.19	2.07	7.97	3.23	11.49	10.40
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
Transfer of the estimated 1996 models to 1991	Wald	1.64	5.31	15.98	4.98	19.92	7.45	26.35	30.61
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59
	LR	0.58	2.19	6.19	2.07	7.97	3.23	11.49	10.40
	$\chi^2_{critical}$	5.99	7.81	9.45	9.49	11.07	11.07	14.07	12.59

Notes: Critical values at 5%.
= null hypothesis of model equality cannot be rejected (model is transferable)

Using commonly used testing procedures in the benefits transfer literature, it can be shown that also DC models extended with these ad-hoc factors are transferable, even though the residual variance in these statistically best fit models is significantly different in the two survey years. Contrary to previous findings, this seems to suggest that the unobserved determinants of preference embedded in the stochastic components of utility over time is not stable in this study. The 1996 model explains less of the variability in the dependent variable than the estimated 1991 model. Hence, important determinants of WTP, which have stayed unobserved, may have been overlooked.

In conclusion, this study suggests that over extended periods real WTP for public goods such as the flood protection and wetland conservation scheme considered here can change by statistically significant amounts. However, the analysis suggests that underlying economic-theoretic determinants of WTP remain stable over such periods. Nevertheless, ad-hoc changes in determinants other than those predicted by theory can result in non-transferability of extended (and statistically best-fit) models. This suggests that transfer exercises might usefully focus upon models with firm theoretical underpinnings rather than incorporating more transitory factors.

2.6.2 Interactive approach

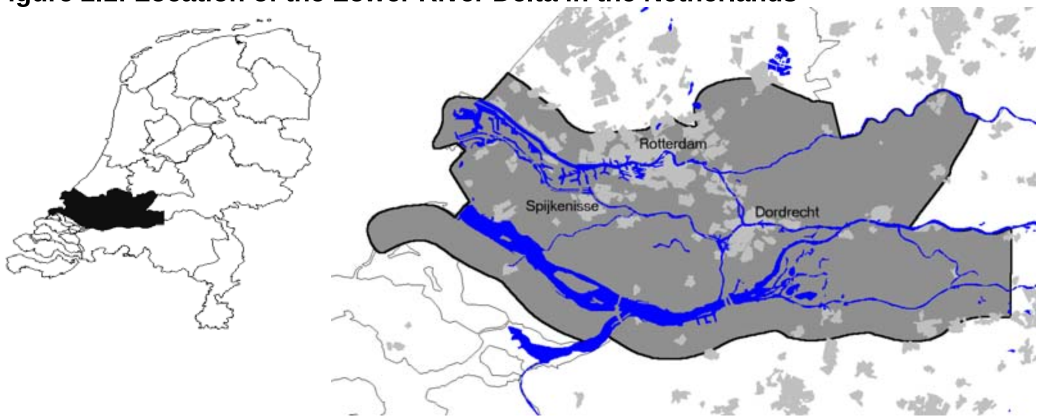
In this second example, the application of benefits transfer will be illustrated with the help of a Dutch case study based on Brouwer and van Ek (2004). For centuries the Dutch have reclaimed and drained land and raised dikes to keep their feet dry. Protection against flooding has always been the Government's primary water policy objective in a country of which approximately two thirds is situated below sea level. Dikes have always been the most important means to achieve this. Since the 1990s Government policy is focusing on alternative ways to maintain existing flood protection and safety levels, such as land use changes in spatial development plans and the restoration of the natural resilience of water systems, including wetlands and floodplains, to absorb excess water.[3] The natural dynamics and flexibility of water systems have been severely undermined in the past through normalisation of rivers, drainage of land and an increase in the built-up area in traditional wetlands and flood plains.

From 1998 until 2000 a Government Working Group investigated in a pre-feasibility study various options for land use changes and floodplain restoration in the Lower River Delta (LRD) along the rivers Lek, Merwede, Meuse and Waal in the Netherlands (

Figure 2.1). The LRA is the estuary of the Rhine and the Meuse in so far as these rivers are influenced by the tides. The critical situations during the floods of 1993, 1995 and 1998 when polders were threatened and tens of thousands of people had to be evacuated prove how topical the danger of flooding is in this region. Awareness is growing that alternative measures have to be taken besides raising dikes to prevent the LRD from flooding in the future.

[3] This new policy is laid down in the fourth National Water Policy Document published in December 1998 and more recently in the recommendations of the Commission looking at important water management issues in the twenty-first century (Commissie Waterbeheer 21^{e} Eeuw) published in August 2000 and the Government's policy paper with respect to these recommendations published in December 2000, titled "*A different approach to water*".

Figure 2.1: Location of the Lower River Delta in the Netherlands



Following the floods in 1993 and 1995, existing dikes were quickly strengthened. However, this measure was largely taken to catch up with necessary maintenance and strengthening of dikes to ensure public safety levels in the short term. To maintain present safety levels and anticipate expected water level rises between twenty centimetres and one metre and fifteen centimetres over the next fifty years (based op different climate change and sea level rise scenarios), alternative land use change and floodplain restoration measures (hereafter referred to as managed realignment) were identified in the area, which provide the same safety levels. These measures will be implemented stepwise between 2000 and 2005, 2006 and 2015, and subsequently from 2016 until 2050. Based on the legally defined safety norms in the area, these measures are part of a planning strategy that is designed to prevent, where possible, new rounds of dike reinforcement and encourage multi-functional use of land and the development of biological diversity present (de Jong et al., 2000). Examples of these measures are shown in Figure 2.2, Figure 2.3 and Figure 2.4.

As part of the Working Group's task, the aggregate effects of these sets of measures in the long term were examined and assessed in detail. Besides an environmental impact assessment (EIA), also an economic analysis was carried out. However, as often is the case, an integrated assessment based on these two separate studies was lacking. The expected impacts of the proposed managed realignment measures are shown in Table 2-6.

Figure 2.2: Deepening of rivers

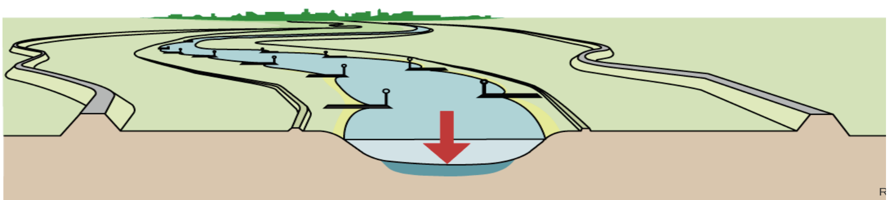


Figure 2.3: Deepening floodplains

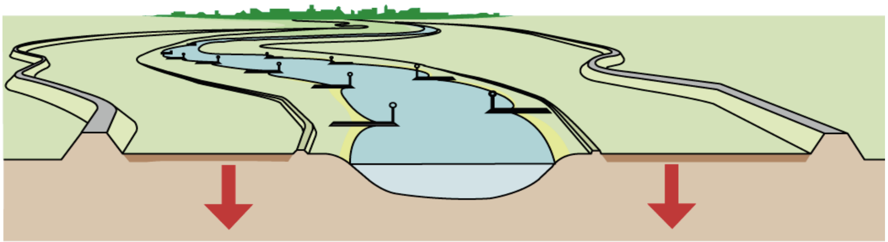


Figure 2.4: Realignment and floodplain restoration

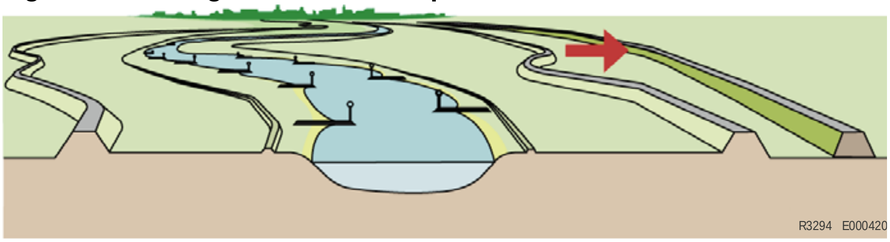


Table 2-6: Expected impacts of managed realignment compared to 'holding the line'

		Priced		***Non-priced***	
		Efficiency	*Redistribution*	*Efficiency*	*Redistribution*
Direct	*Principal Users*	Investment costs Damage costs		Discharge capacity Public perception safety	
	Third parties	Benefits from sand and grit extraction	Income losses in agriculture and industry	- Biodiversity conservation - Public perception dislocation	Employment in agriculture and industry
Indirect		- Recreational benefits - Commercial shipping benefits		Change in water infrastructure	

A distinction is made between priced and non-priced effects, and direct and indirect effects. The most important non-priced positive effects in the case of the proposed managed realignment measures are changes in the water system's discharge capacity, public (perception of) safety and biodiversity restoration. The investment costs needed to implement the managed realignment measures and consequently the damage costs avoided are examples of direct priced effects. The investment costs are borne by the principal who carries out the project (the Government). Important user groups in the region are people who live and own houses in the area, farmers and industry. Their properties and current and future economic interests will be protected by the proposed measures (at the expense of the relocation of a smaller number of houses and businesses). Third parties which benefit from the proposed managed realignment measures are the sand and grit exploitation companies in the area and, consequently, the construction industry, and possibly dredging companies as a result of increased sedimentation.

In view of the positive effects on nature and landscape, the area is expected to become more attractive for recreational activities. The attraction of extra visitors is expected to create more income in the region. These recreational benefits are considered an important indirect effect. The possible effects of the proposed alternative flood control measures on commercial shipping are also indirect effects, which can be relatively easily valued with the help of market prices. The net effect on commercial shipping can be positive or negative. On the one hand, the deepening of river beds and floodplains and the creation of additional water courses is expected to increase commercial and recreational shipping possibilities, while the change in the water infrastructure may also enhance the accessibility of the area. On the other hand, widening the rivers also lowers water levels throughout the river basin, in which case the shipping possibilities decrease.

Another distinction is made between efficiency and redistribution effects. Efficiency effects are included in the economic CBA, while redistribution effects are excluded. Redistribution effects refer to effects which may have important institutional or financial consequences, but which do not influence the economic output of a country, measured in terms of national income or value added. Examples are the loss of income and employment in agriculture and industry in one area or region as a result of the implementation of the proposed land use changes and floodplain restoration measures, which are off-set by income generation elsewhere as a result of the re-location of farms and businesses.

The structure of Table 2-6 is based upon the manual for Cost-Benefit Analysis published in 2000 by the Dutch Ministry of Transport, Public Works and Water Management and the Ministry of Economic Affairs (Eijgenraam et al., 2000). This manual was developed to encourage a more integrated assessment of the various impacts of large infrastructure projects in the Netherlands. Effects which can not be valued in money terms are included, where possible in quantitative terms, in the balance sheet as so-called ‘pro memoriam’ (PM) items. However, in this case, the Working Group’s question was to explicitly value the non-priced social and environmental effects of the proposed alternative flood control measures in money terms in order to assess their effect on social welfare. A preliminary assessment of the economic costs of managed realignment in a cost-effectiveness analysis showed that this option was much more expensive than traditional dike strengthening (holding the line). The total costs of holding the line were approximately £500 million, while the total economic costs of managed realignment were estimated at about £4 billion (Brouwer et al., 2001). The most important reason for these high costs for managed realignment was the fact that the measures are proposed in one of the most densely populated and economically developed areas in the Netherlands with an enormous complex infrastructure, which is expected to be affected significantly by the proposed managed realignment measures.

The Working Group expected that economic (monetary) estimation of the non-priced benefits of managed realignment compared to holding the line might be decisive in concluding whether managed realignment is preferred compared to holding the line. Hence, an important first step was to get the necessary authorisation to carry out an economic valuation study of the main non-priced benefits.

The assessment of the economic value of the expected non-priced social and environmental benefits (public safety and biodiversity restoration) was based on the meta-analysis carried out by Brouwer et al. (1999) based on 30 international studies looking at the economic values of various wetland ecosystem functions (see Table 2-2 in section 2.2). In the Netherlands, no valuation research exists with respect to managed realignment. The 30 studies investigated by Brouwer et al. (1999) produced just over 100 willingness to pay (WTP) values. These values were examined in detail and related to the four main hydrological, geochemical and biological ecosystem functions performed by wetlands: flood water retention, surface and ground water recharge, nutrient retention and export and nursery and habitat for plants, animals and micro-organisms and landscape structural diversity. The economic values associated with these four functions are presented in section 2.2.

The economic values associated with the various wetland ecosystem characteristics are expressed in average willingness to pay (WTP) per household per year. Very often mean values are related to the size of an environmental asset and expressed accordingly, for example in pounds sterling per hectare. This suggests that the average values can be transferred freely and unconditionally over large and small sites irrespective of the number of people who benefit from these sites. An example is the study carried out by Costanza et al. (1997), where based on average values per hectare the total economic value of the world’s ecosystems’goods and services was estimated (see Box 2-2). It is not only the average value used to estimate the value of non-priced environmental benefits which has caused discussion about the ‘right’ prices, also the determination of the ‘market size’, i.e. the number of beneficiaries, has proven to contribute to a large extent to the controversy of using monetary estimates in CBA (e.g. Bateman et al., 2000). Expressing average values per household per year implies that the user of the average values has to think carefully about the exact market size in order to be able to be able to calculate a total economic value, which can be used in the CBA.

The values presented in Table 2-2 in section 2.2 show an average WTP ranging from 18 pounds sterling for the wetland function surface and ground water recharge to 77 pounds sterling for flood water retention. The fact that the function flood water retention is valued highest conforms to expectations considering the possible risks to life and livelihood as a result of flooding and the capacity of wetlands to reduce this risk. No significant difference exists between average values for fresh and saltwater ecosystems. Use values for wetland ecosystems are significantly higher than non-use values (because of the high value attached to flood water retention). Table 2-2 also shows that use and non-use values can not simply be added, as suggested in the literature (Hoehn and Randall, 1989) in order to get a total economic value.

In view of the fact that no valuation results are available in the Netherlands to estimate the economic value of the non-priced benefits of the proposed managed realignment measures, the Working Group agreed to use the values examined in the meta-analysis as the basis for the estimation of a total economic value to be used in the CBA. The results from the meta-analysis were considered the best guesses available. Hence, an important second step was to get the authorisation to use the available information about the estimated economic values of wetland ecosystem functions. The fact that these values were based on not one, but thirty international economic valuation studies, most of which were published in internationally renown journals, is expected to have played an important role in the acceptance of the estimated average values.

The total economic value of the non-priced benefits (i.e. the public perception and valuation of safety, biodiversity preservation and landscape change) is calculated based on the economic values for flood water retention (£77/household/year) and wildlife habitat and landscape diversity (£63/household/year). These values are adjusted for the income differences found between countries, and the fact that use and non-use values can not simply be added. These corrections result in an average WTP for both flood water retention, wildlife and landscape amenities of £53/household/year.[4]

Next, the market size was determined in terms of number of households which are expected to benefit from the proposed managed realignment measures. Together with the Working Group, it was agreed that more or less the whole population of the South-Holland province will benefit. South-Holland contains approximately 1.5 million households. Multiplying this by an average value of £53/household/year results in a total economic value of £80 million per year. Discounted at the prescribed 4% discount rate by the Dutch Treasury over the next 50 years gives a present value of the total economic value of £1.8 billion. The inclusion of this economic value in the CBA still results in a net welfare loss of £900, see Table 2-7.

[4] First, the average values are multiplied by 0.61 (based on estimated regression coefficient) to correct for income differences. Secondly, the income adjusted average values are added and multiplied by 0.62 ([use and non-use]/[use] + [non-use] = 53/85=0.62) to account for the fact that use and non-use values cannot simply be added.

Table 2-7: Present value of costs and benefits of managed realignment in billion pounds sterling (2002 prices)

Costs		**Benefits**	
Investment costs	1.8	Economic risks avoided	0.8
Production loss agricultural land	1.3	Revenues sand extraction	0.3
Maintenance costs	0.7	Economic value public safety and biodiversity preservation	1.8
Total	3.8	Total	2.9

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3 An Instrument for Screening Existing Valuation Studies

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3.1 Introduction

Carrying out environmental valuation studies might sometimes be expensive and time-consuming. An obvious question is therefore whether results from earlier valuation studies can be generalized to new policy settings. For example, could existing results concerning the benefits of an improved water quality in a Polish coastal area be used for saying something about the benefits of such an improvement in a coastal area in France? Such a generalization of valuation results are referred to as benefit transfer, which usually consists of three steps:

1. Identification of environmental valuation studies being potentially suitable for benefit transfer be searching in the scientific and grey literature and/or using databases, among which the Environmental Valuation Reference Inventory (www.evri.ca) is the most comprehensive. There are also smaller and less international databases available, such as the Nordic Environmental Valuation Database (www.norden.org/pub/sk/showpub.asp?pubnr=2007:518), the Australian ENVALUE (www.epa.nsw.gov.au/envalue), and the Swedish ValueBase^{SWE} (www.beijer.kva.se/valuebase.htm). See also McComb et al. (2006) for an overview of international valuation databases.[5]

2. Evaluation of the quality of the studies identified in step #1. This is a very important step in the process. Studies must be screened to identify those which are of a sufficiently good quality to make them suitable for use in benefit transfer. Whilst studies published in peer-reviewed journals might be expected to be of good quality, studies in the grey literature might not have been subject to any quality control. Quality is a multi-faceted feature and it is therefore difficult to create practical quality assessment instruments (QAIs) for valuation studies. One of the few that has been produced is downloadable as a Swedish EPA report from http://tinyurl.com/6phn4p.[6] This QAI is briefly

[5] McComb, G., Lantz, V., Nash, K., Rittmaster, R., (2006). International valuation databases: Overview, methods and operational issues. *Ecological Economics* 60, 461-472.

[6] Söderqvist, T., Soutukorva, Å. (2006) *An instrument for assessing the quality of environmental valuation studies*. Report, Swedish Environmental Protection Agency, Stockholm. Available at: http://tinyurl.com/6phn4p.

described below. A form to be used by an evaluator of a valuation study is available at http://tinyurl.com/5twq62.

3. Transfer of benefits from the studies considered in step #2 to be of acceptable quality. This procedure entails the choice of different transfer methods and their application is an extensive issue which is thoroughly presented in section 2 of this report. The remaining part of this section therefore consists of a description of step #2 only.

3.2 Step #2: Screening existing valuation studies

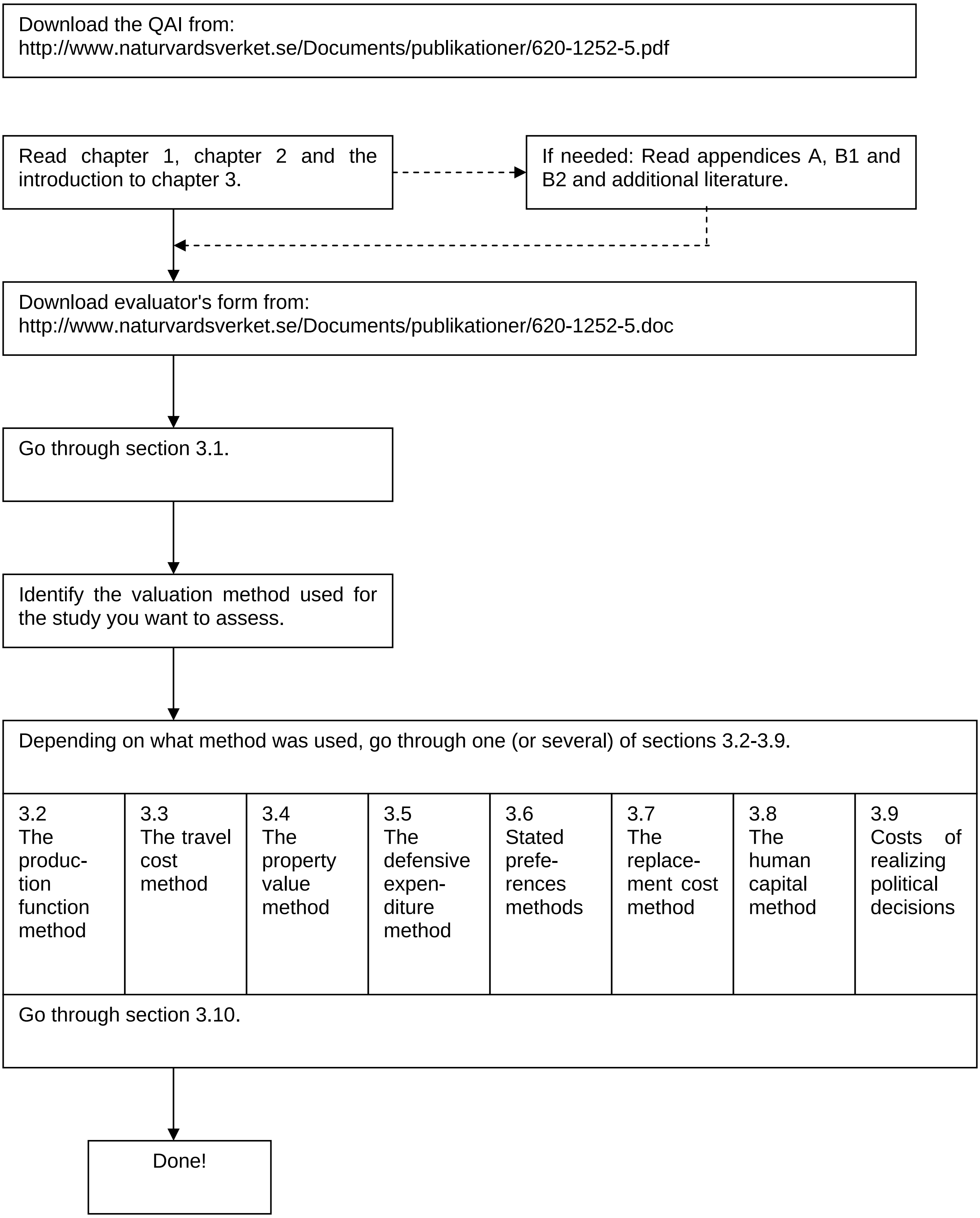
One of the objectives of the QAI in the Swedish EPA report is to communicate crucial aspects of quality to potential evaluators who might not be experts in environmental valuation but have some basic knowledge of environmental economics and statistics/econometrics. This QAI therefore aims at being based on the objectively observable characteristics of valuation studies in order to avoid the kind of subjective assessments that only evaluators equipped with expert knowledge are able to make.

Figure 3.1 shows the procedure for using the QAI. As indicated by the figure, the QAI is based on an identification of a number of factors related to quality for:

a. valuation studies in general, irrespective of what valuation method was used (see section 3.1 in the QAI), and;
b. the application of particular valuation methods (see sections 3.2 to 3.9 in the QAI).

The valuation methods considered in the QAI include revealed and stated preference methods as well as other methods that are less firmly rooted in welfare economics theory. The quality of a valuation study is thus assessed partly through the quality factors in (a) and partly through the quality factors that according to (b) are relevant for the valuation method(s) used in the valuation study. In order to provide an overview, all these quality factors are listed in Box 3-1 and Box 3-2.

Figure 3.1: How to use the QAI



In the QAI, each quality factor is subject to a short discussion, which is followed by one or several check-list questions associated with each quality factor. The purpose of the check-list questions is to make the quality factors more concrete. Most of the questions can be answered by "yes", "no" or "don't know" and they were framed so that "yes" answers are an indicator of good quality. However, "no" or "don't know" answers are not necessarily an indicator of bad quality; this depends on the context and the QAI therefore includes fields for filling in comments that supplement the answers to the check-list questions (e.g. comments about whether a "no" implies a serious weakness of the valuation study or not). Other check-list questions relate to information associated with quality, such as, for example, the non-response rate to a mail questionnaire or interview survey.

Box 3-1: Quality factors for all valuation studies irrespective of valuation method employed

1 Earlier reviews
2 Principal/funder
3 Valuation method
4 Sensitivity analyses related to results from statistical/econometric analyses
5 Are future values discounted?
6 Primary data or secondary data?
7 Data collection
- Survey, population and sample
- The design of the data collection work
- Data collection method
- Non-response
- Survey instrument

8 Access to data
9 Validity test

Finally, the QAI is concluded by an opportunity for an evaluator to give an overall quality assessment, based on the answers to the check-list questions and all other considerations that the evaluator might have (section 3.10 in the QAI). It should be emphasized here that the most important feature of the QAI might not be to find a precise answer to a particular check-list question or to arrive at an unambiguous conclusion on overall quality, but that the QAI simply gives hints to an evaluator on what to look for in a study in order to get an idea of its quality. If no major concerns about the quality of the study arise, it should be safe to proceed to step #3, i.e. the actual benefit transfer procedure. In Söderqvist and Soutukorva (2009)[7], the QAI is applied to two valuation studies, and it might be helpful to have a look at how this was done before applying the QAI for the first time.

[7] Söderqvist, T., Soutukorva, Å. (2009) On how to assess the quality of environmental valuation studies, *Journal of Forest Economics*, 15**,** 15-36.

Box 3-2: Quality factors for particular valuation methods employed

The production function method
1. Natural scientific basis
2. Estimation of changes in producer surplus
3. Modelling of the whole market including dynamic effects

The travel cost method
1. Definition of site(s)
2. Sampling strategies
3. Model specification
4. Calculation of travel costs
5. Opportunity cost of time
6. Multipurpose trips
7. Selection of environmental quality variable

The property value method
1. Property values
2. Property attributes
3. Selection of environmental quality variable
4. Choice and estimation of model

The defensive expenditure method
1. Properties of the good
2. Procedure for estimation of the economic value

Stated preference methods
1. Acceptance and understanding of the valuation scenario
2. Description of effects of the environmental change
3. Information on the null alternative
4. Winners or losers?
5. Payment and delivery conditions
6. Willingness to pay or willingness to accept compensation?
7. Valuation function
8. Test for hypothetical bias
9. Specific quality factors for the contingent valuation method
10. Specific quality factors for choice experiments

The replacement cost method[a]
1. The performance of the man-made system as a substitute
2. The cost-effectiveness of the man-made system
3. Willingness to pay for replacement costs?

The human capital method[a]
1. Theoretical considerations
2. Technological development
3. To estimate the value of lost productivity

Valuation based on the costs of realising political decisions[a]
1. Cost-effectiveness
2. Willingness to pay the costs?

[a] These methods are less firmly rooted in welfare economics than the other methods, but are still included in the QAI because they are often used for environmental valuation.

4 Measuring the Economic Impacts of Tourism and Forecasting Tourism Demand

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4.1 Introduction

Several Spicosa SSAs have highlighted tourism as a key economic issue within their case studies. Indeed it is very likely that tourism will play some role in all Spicosa case studies since almost all coastal areas rely (to a greater or lesser extent) on visitors from other areas for the generation of economic income.

For most SSAs there will be two main kinds of economic issues related to tourism that will arise:

- Establishing the economic impact of tourist activity;
- Forecasting tourism demand, i.e. establishing how numbers of tourists will change in the future under different scenarios.

This note attempts to outline the various ways in which these two issues can be addressed. The emphasis here is to provide simple, practical advice and point readers to a few relevant publications where theoretical and other concerns are described in more detail.

4.2 Measuring the economic impact of tourism

When tourists visit an area the activities they undertake directly or indirectly generate an increase in economic activity within that area; mostly because they increase demand for goods and services. Economic impact studies attempt to measure the economic benefits arising from tourist activity, i.e. the net increase in the wealth of residents (of a country, region, or more specific locality) which is over the levels that would have been achieved without the tourist activity. In layman's terms this means trying to evaluate the changes in sales, income, tax revenues and jobs that come about because of tourism activities. Note that an economic impact analysis differs from a cost-benefit analysis in that the former deals solely with actual flows of money which arise from market transactions whilst the latter includes market and non-market values and tries to assess net social benefit from the perspective of economic efficiency. It is therefore important to understand that any economic impact analysis is a *partial* analysis, in the sense that it does not incorporate all economic impacts, and also since it will be area specific.[8] Note also that

[8] This is important to remember as an increase in economic activity in one area due to tourist activity will be accompanied by a decrease in economic activity in the areas from which the tourists originate. This is worth bearing in mind for case studies where tourists are mostly domestic in origin.

tourism can have negative effects which may be environmental, cultural, social, or economic. These will not be taken into account as part of this kind of analysis.

4.2.1 What are the economic impacts of tourism?

Generally economic impacts are categorised into three types of effects: direct, indirect and induced.

- Direct effects only include the immediate effects of additional demand created by tourism. So, for example, this will include tourism spending on accommodation, meals, recreational activities, and so on.

- Indirect effects relate to the increased demand for goods and services by the industries from which are serving tourists. This would include, for example, the extra food that restaurants need to purchase; the additional inputs of supplies and labour that hotels need in order to cater for tourists, etc. These effects could also include investment in enhanced public transport infrastructure and sewerage facilities, amongst others.

- Induced effects arise when demand for goods and services from households in the region increases as a result of the direct and indirect effects of tourist activity.

4.2.2 Estimating the economic impact of tourism

Most commonly three methods are used to evaluate the economic impact of tourism (or other economic sectors); tourism satellite accounts, input-output (I/O) analysis and computable general equilibrium (CGE) models.

4.2.2.1 Tourism Satellite Accounts

Some organisations and countries have developed national and/or regional sets of tourism satellite accounts (TSAs). These accounts extract information from national economic accounts to try and identify the extent to which tourism contributes to these. These are unlikely to be of use to Spicosa SSAs as they are generally only calculated at a national scale (if they are calculated at all) and it would be difficult to scale down to the more localised areas that are the subject of SSA case studies. Furthermore, TSAs only account for the direct economic impacts of tourism and do not include indirect or induced effects.

Further information on TSAs can be found on the World Travel & Tourism Council's website at:
http://www.wttc.org/eng/Tourism_Research/Tourism_Satellite_Accounting/
where there are also a number of downloadable TSA country reports.

4.2.2.2 I/O Analysis

I/O models describe the flows of money that occur between the different sectors of a region's economy. From these models multipliers can be estimated which measure how much of direct spending is recirculated within the regional economy in terms of indirect and induced effects, thus allowing the overall economic impact of tourist activity to be evaluated.

Many countries produce I/O tables at a national (and some at a regional) level and multipliers derived from these can be adapted for use in SSA case studies. Chapter 5 of this report covers this methodology in much more detail and includes an example of its use as applied to the Clyde SSA. Much more detail can also be found on this methodology (and other tourism economic impact methodologies) from the Department of Community, Agriculture, Recreation and Resource Studies at Michigan State University which has a website devoted to the economic impacts of recreation and tourism: http://web4.canr.msu.edu/mgm2/econ/index.htm.

4.2.2.3 CGE Analysis

CGE models are complex, data intensive models of national or regional economies (see page 55 of D2.1: Economic Assessment for more detail) which can be used to explore the impacts upon an economy of changes in tourism demand. Their complexity and data requirements mean that their use within Spicosa is not possible.

4.3 Forecasting tourism demand

There is a large and growing literature on forecasting tourism demand. Increasingly sophisticated time-series and other econometric models are being employed by researchers, predominately to forecast international tourism demand, i.e. tourist arrivals/departures at/from a particular region or country. The typical form of model that is estimated is:[9]

$$DT_{ij} = f(Y_j, TC_{ij}, RP_{ij}, ER_{ij}, QF_i)$$

Where;

DT_{ij}	=	demand for international travel services by origin *j* for destination *i*;
Y_j	=	income of origin *j*;
TC_{ij}	=	transportation cost between destination *i* and origin *j*;
RP_{ij}	=	relative prices (i.e. the ratio of prices in destination *i* to prices in origin *j* and in alternative destinations);
ER_{ij}	=	currency exchange rate between destination *i* and origin *j*;
QF_i	=	qualitative factors in destination *i*. A large number of qualitative factors can effect demand for tourism travel and can include; gender, age, education level, household size, destination attractiveness, political, social and sporting events in the destination, etc.

Economists have certain expectations of how some of these variables might be related to demand for travel to a particular destination. For example, a positive relation is expected between demand and income in the origin

[9] This example is taken from Lim, C. (2006) A survey of tourism demand modelling practice: issues and implications, in, Larry Dwyer and Peter Forsyth (Eds.) *International Handbook on The Economics of Tourism*, Edward Elgar, Cheltenham.

country and a negative relationship is expected between relative tourism prices and transportation costs.

There are two ultimate aims of these models;

1. to estimate trends in demand and extrapolate these into the future;
2. to estimate parameters for each of the variables included the model which will allow calculation of elasticities.

An elasticity is economic jargon for a simple concept. For our example the income elasticity of tourism demand for travel to destination *i* refers to the ratio of the percentage change in tourism demand for travel to destination i to the percentage change in income in origin *j*. So if demand changes at the same rate as does income then the elasticity estimate will have a value of 1 (unitary elasticity). Different types of goods have different elasticities. Demand for necessities, such as staple food items (e.g. bread, rice), are usually inelastic with respect to their own price (e.g. a 1% rise in price will cause a less than 1% fall in demand). Luxury goods, such as tourism, are generally elastic with respect to their own price (e.g. a 1% rise in the price of travel to destination *i* will cause a greater than 1% fall in demand). Although a simple concept these elasticity estimates can provide very useful information to policymakers and researchers who are interested in how tourism demand changes in response to income or prices or other variables.
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4.3.1 Practical implementation of tourism forecasting methods within Spicosa

As previously mentioned these kinds of models are estimated using a variety of times-series and econometric techniques of varying degrees of sophistication. Since all estimations are made using statistical techniques, the statistical significance of the estimates obtained from the model increases as the number of observations increases. Hence most research in this area has used data sets based on national data which has been collected according to consistent data definitions and which is (generally) available over a number of years. For this reason it is unlikely that SSAs will themselves be able to estimate demand for tourism for their case study areas since there will not be enough good quality data available to them that is specific to the geographical area they are dealing with. The most practical course of action open to them will be to use estimates of overall trends or elasticities for specific variables from existing studies.

4.3.2 Tourism demand forecasting studies

There are a large number of studies that have been done related to estimation of tourist demand to or from certain areas. The majority of these use data on movements of tourist between countries and there are few that model demand for domestic tourists. It has not been possible to do a comprehensive literature search to find studies specific to each country represented within Spicosa and so the listing of publications that follows only includes general papers which are either meta-analyses or reviews of tourism demand studies.

These papers give further details of the approaches used. It is recommended that if SSAs wish to use parameters estimated from such models then they should carry out a country specific literature search in order to find literature that would be relevant to their study site.

Crouch, G. I. (1995) *A meta-analysis of tourism demand*. Annals of Tourism Research 22**,** 103-118

Eilat, Y. and Einav, L. (2004) *Determinants of international tourism: a three-dimensional panel data analysis*. Applied Economics 36**,** 1315 – 1327

Lim, C. (1997) *Review of international tourism demand models*. Annals of Tourism Research 24**,** 835-849

Smeral, E. and Weber, A. (2000) *Forecasting international tourism trends to 2010*. Annals of Tourism Research 27**,** 982-1006

Song, H. and Li, G. (2008) *Tourism demand modelling and forecasting - A review of Recent research*. Tourism Management 29**,** 203-220

Witt, S. F. and Witt, C. A. (1995) *Forecasting tourism demand: A review of empirical research*. International Journal of Forecasting 11**,** 447-475

The World Tourism Organization (UNWTO) has also published more general forecasts for tourism demand for Europe in its 2001 publication *Tourism 2020 Vision Vol. 4 Europe*, although the accuracy of these forecasts is likely to have been adversely affected by the recent economic downturn. The UNWTO website also has a variety of other information and publications available that may be of some use: http://www.e-unwto.org.

4.3.3 Tourism demand and climate change

Climate change is very likely to have a significant impact on patterns of tourism demand in Europe over the long-term and this should be acknowledged in all SSAs where tourism is of importance. Possible effects of climate on tourism have been explored in the following recent journal articles:

Bigano, A., Bosello, F., Roson, R. and Tol, R. (2008) *Economy-wide impacts of climate change: a joint analysis for sea level rise and tourism*. Mitigation and Adaptation Strategies for Global Change 13**,** 765-791

Bigano, A., Hamilton, J. M. and Tol, R. S. J. (2006) *The impact of climate on holiday destination choice*. Climatic Change 76**,** 389-406

Hamilton, J. M., Maddison, D. J. and Tol, R. S. J. (2005) *Effects of climate change on international tourism*. Climate Research 29**,** 245-254

Hamilton, J. M. and Tol, R. S. J. (2007) *The impact of climate change on tourism in Germany, the UK and Ireland: a simulation study*. Regional Environmental Change 7**,** 161-172

See also for a comprehensive bibliography on climate change and tourism and recreation:

Scott D., Jones B. and McBoyle G. (2006) Climate, Tourism and Recreation: A Bibliography - 1936 to 2006.which is available to download at: http://www.fes.uwaterloo.ca/geography/faculty/danielscott/PDFFiles/CTREC%20Biblio_June%202006.pdf.

Information can also be found on the eCLAT (Experts in Climate Change and Tourism) website at: http://www.e-clat.org/.

4.4 Summary

All methods for evaluating the economic impact of tourism and for forecasting tourism demand are data intensive and complex, and for these reasons are probably beyond the resources available to SSAs. However, information provided by these methodologies – in the form of multipliers, or elasticities – could be extracted from existing studies and used by SSAs. Note, however, that this should only be done if the economic conditions at the study site conform reasonably closely to the economic conditions that existed at the time and place of the original study. Additionally, sensitivity analysis should be used to determine how important these transferred parameters in the context of the final model results.

5 Note on the use of Input-Output Multipliers in economic impact assessment

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5.1 Introduction

The purpose of this brief note is to illustrate the use of output multipliers to assess the regional economic benefits from an economic activity. Those multipliers can be used to assess the impact of all types of economic activity but seem particularly useful for service industries (tourism activity is used as an example). In this note, an illustration of the practical use of multipliers for economic impact assessment of tourism (using the example of SSA 7 Firth of Clyde) will be followed by the theoretical description of the Input-Output - hereafter I/O- analysis framework and of the calculation of multipliers.

Please note that, besides the use of the output multipliers described here, an I/O framework can also help to assess:

- The impact of policy options/environmental changes on production levels, employment or price levels in other sectors (on the economy in general), i.e. indirect impacts of environmental policies
- If it is "greened", the impact of policy options on the level of pollutant emitted (for a given sector or in total in the region)[10].

It seems particularly suitable to take into account the *indirect impacts* of environmental policies and/or environmental changes. For a description of the other regional accounting methodologies that might be of use for SSAs, please refer to D2.1.

5.1.1 Economic impact assessment of the tourism industry

I/O methodology is particularly suitable for the evaluation of regional services industries (e.g. tourism) and the impact assessment of broad policy instruments at the regional level (Sun, 2007). Since tourism is an important issue within SPICOSA, the main part of this note will focus on the impact assessment of tourism sector activities. The steps towards an economic impact assessment will be described. General considerations will be followed by a concrete example: Firth of Clyde study site being used as an illustration.

Please bear in mind that, as Stynes (1997)[11] points out, economic impact analysis only helps answer the question "**What is the contribution of**

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[10] If you are interested in this type of assessment, please refer to the available example on WP2 FTP.

[11] Please refer to his bulletin for a more complete and contextual presentation of economic impact assessment of tourism.

tourism activity to the economy of the region?" (in terms of changes in income and employment in the region). To get a whole picture of the impacts of tourism, an environmental impact assessment[12] or a cost benefit analysis should be implemented as well: while the economic impact analysis tends to emphasize the positive benefits of tourism, environmental, social or cultural impact studies tend to highlight the negative consequences of tourism.

5.1.1.1 Economic impacts of tourism

The value of tourism is not restricted to the direct activity supported by tourism-related expenditure (the so-called *direct effect*) (see the example in the box below). Indeed, industries directly supported by tourism spending such as hotels and restaurants have backward linkages (purchasing links) to other firms in a region. The level of these links can vary, depending on the integration of the tourism-related industries in the reference region. As tourism-related sectors purchase goods and services in the regional economy they then support additional output, employment and incomes in supplying sectors: this is the so-called *indirect effect*.

In addition, there are a series of *induced income effects* as those employees whose jobs are supported in the value chain spend their incomes on regional goods and services thereby supporting other economic activity.

> **Example:** Stynes (1997, p 4-5)
> "Let's say a region attracts an additional 100 tourists, each spending $100 per day. That's $10,000 in new spending per day in the area. If sustained over a 100 day season, the region would accumulate a million dollars in new sales. The million dollars in spending would be distributed to lodging, restaurant, amusement and retail trade sectors in proportion to how the visitor spends the $100. Perhaps 30% of the million dollars would leak out of the region immediately to cover the costs of goods purchased by tourists that are not made in the local area (only the retail margins for such items should normally be included as direct sales effects). The remaining $700,000 in direct sales might yield $350,000 in income within tourism industries and support 20 direct tourism jobs. Tourism industries are labor and income intensive, translating a high proportion of sales into income and corresponding jobs.
> The tourism industry, in turn, buys goods and services from other businesses in the area, and pays out most of the $350,000 in income as wages and salaries to its employees. This creates secondary economic effects in the region. The study might use a sales multiplier of 2.0 to indicate that each dollar of direct sales generates another dollar in secondary sales in this region. Through multiplier effects, the $700,000 in direct sales produces $1.4 million in total sales. These secondary sales create additional income and employment, resulting in a total impact on the region of $1.4 million in sales, $650,000 in income and 35 jobs. While hypothetical, the numbers used here are fairly typical of what one might find in a tourism economic impact study."

To sum up, this means that initial rounds of tourism expenditure will have a series of direct, indirect and induced economic effects and that the total

[12] Weisskoff (2000) provides a good example of combined I/O and environmental analysis.

economic impact is the sum of these effects within a region. Put another way, the direct effects are *multiplied* up to derive the total economic effects of spending because of value chain (indirect) and induced income effects. This can be taken into account while using the I/O framework and **Input-Output multipliers** that capture the secondary economic effects of tourism activity[13].

5.1.1.2 How to measure the economic impacts of tourism[14]

Basically, the economic impacts of tourism are estimated using the simple formula:

Economic impact of tourism =
Number of tourists * Average spending by visitor * Multiplier

This formula suggests 3 steps for estimation:

(1) Estimation of the change in the number (and perhaps type) of tourists to the region

Estimates or projections for the number of tourists in an area are not always available. Some local reports on the levels of tourism activity might solve this data need (this corresponds to primary data).
Demand models might also be used, provided they are enough disaggregated (this might be considered as secondary data in a more or less disaggregated form).
Good judgment can also help. In the absence of any real data, you would have to rely on subjective estimates.

Within SPICOSA, it seems that it is mostly scenarios on the increase of the number of local tourists that will be explored and different trends of tourist flows assessed, relative to their economic as well as on their environmental impact. However, the "baseline trend" with respect to the number tourists would also be useful to consider and estimate, in order to be used as comparison.

(2) Estimation of the average levels of spending of tourists in the area

The I/O approach enables the assessment of the effects of individual developments, provided that they can be linked with a change in tourism consumption spending. **The final demand change** (change in sales to the final consumers of goods and services) **needs to be estimated.**

One thus needs to estimate the **mean expenditure** of a tourist at local businesses. When no visitor spending survey exists for the area you are studying, such a survey might need to be implemented (this means acquiring primary data). Those estimates

[13] These multipliers can be of Type I or Type II and relate to output, income or employment: read the theoretical considerations of section 2.
[14] The structure and contents of this section mainly builds on Stynes (1997).

should be based on a representative sample, taking into account variations across seasons, types of tourists and locations within the studied area. Spending averages might also be borrowed or adapted from other studies: it is possible to rely on secondary data, adjusting it over time using consumer price indices. As for the estimation of the number of tourists, expert judgment or guesstimates might also be used.
Within SPICOSA it generally seems to be the case that only segments of the total tourism industry will be explored and assessed. However, estimation of the average level of spending should, as much as possible, take into account the differences existing within the tourist segment studied: i.e., local residents vs. tourists, day users vs. overnight visitors, etc. and consider different spending patterns.

When the number of tourists is multiplied by the average spend per visitor, one gets an estimate of the total tourism spending in the area or **direct effect**, which represent the amount of money brought into the region by tourists.

(3) Estimation of the local I/O multipliers to determine secondary effects of tourism

To estimate the **secondary effects** of tourism spending, the I/O multipliers can be used. Multipliers can come from a regional I/O table or be re-scaled from a national I/O table (this can be considered as secondary data). It might be more effective if sector-specific multipliers are used, rather than aggregate tourism spending multipliers.

Care should be taken if you wish to transfer multipliers from existing studies (secondary data), in other contexts: one should not take a multiplier adjusted for one region and apply it in a region of quite a different economic structure.

Stynes (1997, p. 7) notes that" to properly apply tourist purchases of goods to an I/O model (or corresponding multipliers) various margins (retail, wholesale and transportation) must be deducted from the "purchaser price" of the good to separate out the "producer price". Indeed, in an I/O framework, retail margins accrue to the retail trade sector, wholesale margins to wholesale trade, transportation margins to transportation sectors (trucking, rail, air etc.) and the producer prices of goods are assigned to the sector that produces the good". Moreover, he states (p. 7) that "in most cases the factory that produces the good bought by a tourist lies outside of the local region, creating an immediate "leakage" in the first round of spending and therefore no local impact from production of the good. Before applying a multiplier to tourist spending, one must first deduct the producer prices of all imported goods that tourists buy (i.e. only include the local retail margins and possibly wholesale and transportation margins if these firms lie within the region). Generally, only 60 to 70% of tourist spending appears as final demand in a local region".

If uncorrected or rough tourism spending is used, the calculations are thus bound to generate an inflated estimate of tourism impacts. However, such a corrected value for tourism spending is difficult to properly estimate.

5.1.2 Data availability: Input-Output Tables

Input/Output tables are available for most of the European countries (except Romania and Cyprus) and for Norway and Turkey.

To access the workbooks by country you can go to http://epp.eurostat.ec.europa.eu/portal/page?_pageid=2474,54156821,2474_54764840&_dad=portal&_schema=PORTAL#IOT.[15]

It might also be useful to investigate if regional authorities provide regional tables, more suited to the studied area.

When you have the data, the I/O multipliers can be calculated, using the methodology described in the second section of this document. Note that for the calculation of the I/O multipliers, all the information available (all the industries covered in the table) will be needed. However, if relevant for the issue you wish to study, some categories of expenditure might be aggregated by sectors (in the same way as in the example for the Netherlands below).

5.1.3 Disaggregating and re-scaling the multipliers

To use the multipliers to assess the economic impact of the chosen tourism issue, you will first need to identify the most relevant categories for the studied issue (select the categories of expenditure in the I/O tables). This part of the assessment can be problematic since tourism employment and activities are difficult to precisely define: visitors often use other services than restaurants and hotels such as postal or health services and tourism is rarely a homogeneous activity (Jones and Munday, 2004).

In order to disaggregate the multipliers to refine the analysis, the national or regional multipliers often need to be re-scaled to a local level. It would be an error to apply a statewide multiplier to a local region, since multipliers tend to be higher for larger regions with more diversified economies and hence their use would yield estimates of local multipliers effects which would be biased upwards.

The use of an I/O framework to assess tourism activity is not always clear-cut. Data on the tourism industry at a regional level is often scarce. You can rely

[15] If you want some precise data on a particular industry or product, click on "Database functions" and Select in the folders "Economy and finance", "National accounts (including GDP)", "Supply, use and Input-Output tables", "Tables at current prices", "Input-output table-current prices". The table can then be downloaded once you have made your selection regarding the sector, country and year you want to study.

on rough estimates of the scale of tourism to disaggregate the multipliers. However, if you wish to refine the regional Input-Output table, it might be necessary to collect statistical information on the size and transactions of industries with a significant degree of tourism dependence. A regional survey on tourism activities (business activities, employment, purchasing, etc.) is thus sometimes needed. (Jones and Munday, 2004).

5.1.4 Example of economic impact assessment: SSA7 Firth of Clyde[16]

In general it seems important to **clarify the nature of the problem being addressed** before launching an economic impact study. In the Firth of Clyde study site, the local stakeholders were interested in “the implications of increased leisure and tourist use of the Firth of Clyde”, from the perspective of combining increased tourism trends with social, economic and environmental stability. The socio-economic issue of the SSA has been focused on the implications of increased leisure and tourist use of the Firth of Clyde, in particular for recreational boating activities, but also for mussel farming. However, only the tourism part will be the focus here[17].

The action to be valued is thus the increase in quantity of tourist facilities due to increasing tourism trends. McKensie Wilson (2006), studying the potential for development of the sailing industry in the Clyde, suggested that the whole Clyde estuary could double its berthing capacity for recreational boating by 2015, in response to an increased demand.

The **change in number of tourists** will be assessed through several scenarios: a 50% increase by 2013 of marine recreation activities, a 50 to 100% increase by 2013 of the area available for aquaculture and a proposal of Marine Protected Area to cover 20% of Firth of Clyde. Evaluating this range of alternatives of tourist trends will also help to evaluate the sensitivity of the results[18].

For the moment, only the baseline scenario (with no increase with respect to the current situation of recreational boating) has been formulated in Extend. The number of boats in the marina with respect to the season has been introduced in a lookup table. **Mean spending** per boat per stay has been estimated to be £250, with an average length of stay of 3 days.

These data could easily be refined since precise data on this tourist segment exists for the whole Clyde Estuary, even if these data need to be rescaled to fit the chosen boundaries of the system. McKenzie Wilson (2006) report very precise figures about annual expenditure associated with sailing activity in the Clyde. They also make a distinction, in the spending patterns, between

[16] More examples of economic impact assessment of tourism are available in, for instance, Stynes (1999).
[17] Although the Scottish team is also using I/O for impact assessment of mussel farming.
[18] If only one change in activity is assessed and is subject to uncertainty, the impacts could be evaluated using a range of estimates in order to establish rough confidence intervals around these estimates.

different types of tourists (permanent, seasonal berth holders, visiting boats, etc.). Moreover, the report also focuses on the expenditure inside the Clyde area, which is important to make the assessment of the secondary effects relevant (although they will be overestimated if multiplied directly by the multipliers since the available data use “purchaser prices” and not “producer prices” –see above–).

The **direct effects** of the increase in tourism trends are captured by these estimates of visitor spending. They can be completed by current employment estimates in the sector or estimations of income and profits for instance.

In order to get a better estimate of the economic effects of increased recreational boating in the area i.e. **evaluate the secondary effects** of sailing expenditure in the Firth, output multipliers are used. Fortunately, the Scottish Executive provides regional Input-Output tables as well as Type I and Type II multipliers[19]. Thus, in this case, the I/O multipliers do not need to be calculated and data can be found at a more regional level than on Eurostat (even though these data need to be further disaggregated to be suitable for a local analysis, at the Firth of Clyde level).

The McKenzie Wilson (2006) report helped identify in the Scottish multipliers table (in which expenditure is highly disaggregated) 4 **key economic sectors** that are mainly impacted by the recreational boating activities (they represent relevant categories of spending). These categories and their multipliers are shown in Table 5-1.

Table 5-1: Type II multipliers for Scottish economic sectors impacted by recreational boating

	Scotland output multiplier (Type II)	Estimate of the Clyde area multiplier (Type II)
Recreational, cultural and sporting activities	1.93	1.7
Tourism (hotels, catering and pubs)	1.49	1.4
Supporting and auxiliary transport activities	1.30	1.2
Retail	1.71	1.5
Average	1.61	1.45

Source: McKenzie Wilson (2006) : *Sailing in the Clyde economic impact study*, Scottish Enterprise Ayrshire, Glasgow.

The *local* (at the Clyde Estuary level) **output multipliers** for these sectors are not available and the actual values will depend on the structure and linkages of businesses and industry within the area. McKenzie Wilson (2006) estimate

[19] Input-output tables and multipliers for Scotland, Scottish Executive, 2005 (2002 data), available online: http://www.scotland.gov.uk/Topics/Statistics/Browse/Economy/Input-Output.

that the indirect effect in the Clyde area is 75% of the Scottish indirect effect[20] (to our knowledge, this figure is not especially based on a survey, but is mostly a rough estimate) and assume an equal spend across the 4 categories to calculate the average and get a unique multiplier. When such disaggregated data is available, one may use this type of average multiplier -if a general measure of impact is desired. One may also calculate estimates of income and jobs broken down by sector to get a more detailed estimation.

These figures will need to be further disaggregated in order to get sector and average output multipliers corresponding to the Firth of Clyde, which is only a part of the whole Clyde area on which the McKenzie Wilson (2006) report is based.
To do this, one could refer to the infrastructure available in the Firth (number of marinas, pontoons, moorings, etc), compare it to the rest of the Estuary and use the information available to scale down the multipliers.
Anyway, within the SPICOSA project, the output multiplier figures can be refined later on (during the Appraisal Step), which leaves some time to gather the data/information needed to further re-scale the multipliers.

The Scottish team also uses **employment effects and multipliers**, which can be rescaled from the Scottish table using the same techniques as for the output multipliers. These should be used and interpreted carefully. Stynes (1997) indeed recommends income or value added as the best measures of economic impact to report. Job impacts might, according to him, be misleading because jobs in the tourism sector are largely part-time and seasonal. Wages and salary rates vary as well across industries, which can make the multipliers vary accordingly.

Since this type of economic impact assessment only implies the use of a multiplication function, the **formulation in Extend** of the use of I/O multipliers is quite clear-cut.
The I/O framework does not explicitly keep track of time but it is commonly assumed that the impact estimates represent activity within a single year. This might cause some trouble with the time step of the model because one should sum tourist expenditures for one year and multiply yearly by the multiplier. Solutions to this problem do exist but still have to be tried out by the Scottish team.

5.1.5 Discussion and limitations of the I/O method – to go further

- The technical coefficients used above are static. It is possible, although data demanding, to calculate dynamic technical coefficients to integrate progress in the production structure. Stynes (1997) argues, amongst others, that Input/Output coefficients can be assumed stable during a certain period after the initial calculation of the I/O table (typically up to 5 years), unless the region's economy has changed significantly.

[20] For "Recreational, cultural and sporting activities" this represents 75% of 0.93 (indirect and induced effects) + 1 (direct effect, resulting from the change in final demand, increase in one unit of the output in the local economy): 0.95*0.75 + 1 = 1.7.

- The level of error in the results should be controlled for. The errors could come from the three components that are estimated in the analysis: visits, spending and multipliers. As Stynes (1997, p. 13) states, “The more accuracy you demand, the greater requirements to gather up-to-date local data on visitation, spending and economic activity”. A trade-off thus needs to be made between time, money and knowledge required to fine tune the estimates and the benefits of their improvement.

  To further study the level of error, a **sensitivity analysis**[21] can also be used and the coefficients re-calculated: either *absolute or relative changes* can be examined. The absolute change can for instance increase/decrease each element of the domestic intermediate matrix by 10%; this absolute change does not change the relative importance of each expenditure since it is distributed proportionally to the column elements. However, if the column sum of the domestic intermediate matrix is increased/decreased by, for instance, 0.10, the importance of each expenditure would be affected (that is why it is called relative change). Since an increase/decrease in the elements would imply that total expenditures are greater/less than total receipts, these changes need to be accounted for by using import substitution (increasing or decreasing imports).

  The sensitivity analysis is based on estimating the differences between the original output multipliers and the multipliers estimated using the absolute and relative changes[22].

- One may wonder if all additional activity should be treated as a net benefit since some of the elements of expenditure might be displaced from elsewhere.
  Indeed, as Stynes (1997) underlines, multiplier computations for induced effects generally assume that jobs created by additional spending are new jobs, involving new households in the area. Induced effects are computed assuming linear changes in household spending with changes in income. Estimates of induced effects might be inflated due to the violation of these assumptions. However, there might be no solution to correct the estimates and take these displacement effects into account (Jones and Munday, 2004).

- Some authors also argue that total output and factor inputs are strongly influenced by capacity utilization[23] which makes the assumption of stability of ratios and multipliers flawed. Indeed, if demand grows faster than supply, utilization rates will increase and standard I/O models lead

[21] This section follows Wagner (1997), though he does apply this sensitivity analysis for a Social Accounting Matrix.
[22] According to Wagner (1997), if the mean difference for any submatrix is respectively : less than the population mean ± 1SD; bounded by the population mean ±1SD and ± 2SD; greater than ± 2Sd, then the output multipliers are respectively considered to be: not sensitive to the changes; slightly affected by the changes and readily affected by the changes.
[23] Defined as the ratio of actual used products to the total available products.

to an overestimation of jobs and personal income and an underestimation of profits (and the other way around when it utilization rates decrease). Sun (2007) proposes a model to take into account capacity utilization and modify I/O ratios and multipliers. However, it is technically difficult to implement.

To conclude, **bear in mind**, at all times, **the assumptions and limitations of I/O while analyzing and using the results**, do not oversimplify or misinterpret the results, use them with caution, since it might leave the stakeholders with a sometimes distorted or incomplete understanding of tourism economic effects.

It might be useful to provide a range of values, rather than a single figure as estimate of the economic impacts. It is also recommended to report both the direct and the indirect effects, in order to present a good picture of the assessment.

5.2 Basic principles of Input-Output Methodology and calculation of Input-Output Multipliers

An Input-output matrix (I-O) is a **representation of national or regional economic accounting** that records the way industries trade with one another and produce (flows of products and services). Those flows are registered in a matrix, simultaneously by origin and by destination (OECD, 2006). The matrix illustrates the relationship between producers and consumers as well as interdependencies of industries for a given year.

Table 5-2 shows the basic structure of the I-O table (also called the transaction matrix). Basically, the rows represent the outputs (suppliers) and the columns the destination of inputs (users).

Example[24].
If we look at the section called the domestic intermediate matrix (in red in Table 5-2), 6 sectors are represented: Agriculture with an annual production (or output) of 21863, Mining producing 12292, Manufactures with a total output of 210900 and Utilities; Construction and Services producing respectively 18249, 60244 and 435953. All those outputs are read on the row/column "industry inputs at basic prices" (sum of inputs and outputs are typically equal[25]) and are in million Euros 2000[26].

All the cells of the domestic intermediate matrix show the flows between sectors.

[24] The example developed throughout this second section builds on the 2000 Input-Output table for The Netherlands, available in the workbooks of Eurostat:
http://epp.eurostat.ec.europa.eu/portal/page?_pageid=2474,54156821,2474_54764840&_dad=portal&_schema=PORTAL
[25] Since the columns represent the destination of inputs and the rows sum to total output of a sector, the matrix represents a national/regional approach to double entry bookkeeping; total input and output of a sector are equal to each other.
[26] For a more detailed description of the different sections of the matrix, please refer to D2.1.

Example, *cont.*
If we take the *rows*, for Agriculture: of the 21863 million Euros produced, 2731 have been used inside the sector itself, whereas respectively 3, 8260, 36, 59 and 615 million Euros have been used in, Mining, Manufactures, Utilities, Construction and Services.
The *columns* describe the structure of the input of the corresponding sector. For instance, to produce 21863 million Euros, Agriculture needed 2731 of its own production, and, respectively, 4, 3322, 983, 121 and 2884 of the production in Mining, Manufactures, Utilities, Construction and Services.

It is important to note that I/O tables assume linear relations between inputs and outputs from different sectors as well as linear relations between outputs and final demand. This assumption is not always close to reality: it means that there are no economies or diseconomies of scale in production or factor substitution (double the level of production, you'll need to double all the inputs). Moreover, all firms in a given industry are assumed to employ the same production technology.

Table 5-2: Domestic transactions input-output table (in million Euros 2000)

	Agriculture, forestry and fishing	Mining	Manufactures	Utilities	Construction	Services	Private final consumption	*of which*, final consumption expenditure by households	Governement final consumption	GFCF	Exports	Industry outputs *at basic prices*
Agriculture, forestry and fishing	2731	3	8260	36	59	615	962	*962*	62	567	8568	21863
Mining	4	282	2013	3979	188	60	28	*28*	0	210	5528	12292
Manufactures	3322	291	40218	480	8004	16999	16896	*16896*	2340	8573	113777	210900
Utilities	983	53	2400	4395	85	3458	6184	*6184*	14	439	238	18249
Construction	121	70	565	135	14103	9509	405	*405*	530	33974	832	60244
Services	2884	1078	28400	1404	9339	106994	126180	*123398*	87409	16752	55512	435953
Imports	1779	1029	71117	1878	7572	33964	*24189*	*24189*	*1085*	*17771*	*81863*	
Net taxes on products	129	67	497	706	249	8651	*22908*	*22954*	*-152*	*10233*	*0*	
Total Use *at purchaser's prices*	11953	2873	153470	13013	39599	180250	*197752*	*195016*	*91288*	*88519*	*266318*	
Compensation of employees	*2336*	*520*	*35083*	*1629*	*14339*	*151784*						
Value Added *at basic prices*	9910	9419	57430	5236	20645	255703						
Industry Output *at basic prices*	21863	12292	210900	18249	60244	435953						

Source: OECD (2006): OCDE aggregation of 2000 input-output table for The Netherlands and own calculations based on Eurostat

Another section called domestic investment matrix (in blue in Table 5-2), accounts for the supplies of goods that are not consumed by domestic industries. The columns therefore include such categories as final consumption (both by households and general government), gross fixed capital formation (investment) and exports.

The initial monetary values in the transactions matrices can be converted into ratios called **technical coefficients** (Table 5-3). This is done by dividing each cell of the domestic intermediate matrix by its column total (output at basic prices).

Example, *cont.*
In the first column-third row, the technical coefficient is equal to 3322/21863 = 0.15. This coefficient shows the rate at which inputs are transformed into outputs. Here, 0.15 Euros manufactured products are purchased by Agriculture in order to produce 1 Euro of agricultural output.

These technical coefficients can be used directly to assess the added value of a given sector, calculate investments in this sector and, if the matrix is "greened" [27], determine the impact of this sector on the level of pollutant emitted. However, this type of use of I-O analysis is not the focus here. If you are particularly interested in this type of study, please refer to the example available on WP2 FTP.

In order to finally calculate the **output multipliers**, one needs to derive Leontief inverse matrices.

The **type I inverse matrix** shows how much of each industry's output is needed, in terms of direct and indirect requirements to produce one unit of a given industry's output. It is calculated using the formula:

$$L = (I-A)^{-1}$$

Where
L is the Leontief Inverse matrix
I is the Identity matrix
A is the Direct Requirements matrix (each cell of the domestic intermediate demand quadrant divided by its column total i.e. square matrix of technical coefficients)

Example, *cont.*
When the technical coefficients have been calculated (shaded area in Table 5-3), an identity matrix of the same dimensions as the so called direct requirements matrix needs to be constructed (in this case the dimension is 6*6, see Table 5-4). The A matrix then needs to be subtracted from identity matrix to produce the "I-A" matrix (Table 5-5). This "I-A" matrix must be inverted to construct the type I Leontief inverse matrix (Table 5-6). All these basic matrix calculations can easily be performed in spreadsheets (for instance in Excel[28]) or in other programs.

[27] A line on pollutant production by each sector and a column on the production of goods and services resulting from the implementation of environmental measures is added to the classical I-O table.
[28] At least for a matrix of dimensions 52*52 in Excel.

Table 5-3: Technical coefficients

	Agriculture, forestry and fishing	Mining	Manufactures	Utilities	Construction	Services	Private final consumption	*of which*, final consumption expenditure by households	Governement final consumption	GFCF	Exports	Industry outputs *at basic prices*
Agriculture, forestry and fishing	0,12	0,00	0,04	0,00	0,00	0,00	0,00	0,00	0,00	0,01	0,03	21863
Mining	0,00	0,02	0,01	0,22	0,00	0,00	0,00	0,00	0,00	0,00	0,02	12292
Manufactures	0,15	0,02	0,19	0,03	0,13	0,04	0,09	0,09	0,03	0,10	0,43	210900
Utilities	0,04	0,00	0,01	0,24	0,00	0,01	0,03	0,03	0,00	0,00	0,00	18249
Construction	0,01	0,01	0,00	0,01	0,23	0,02	0,00	0,00	0,01	0,38	0,00	60244
Services	0,13	0,09	0,13	0,08	0,16	0,25	0,64	0,63	0,96	0,19	0,21	435953
Imports	0,08	0,08	0,34	0,10	0,13	0,08	*0,12*	0,12	*0,01*	*0,20*	*0,31*	
Net taxes on products	0,01	0,01	0,00	0,04	0,00	0,02	*0,12*	0,12	*0,00*	*0,12*	*0,00*	
Total Use *at purchaser's prices*	0,55	0,23	0,73	0,71	0,66	0,41	*1,00*	1,00	*1,00*	*1,00*	*1,00*	
Compensation of employees	0,11	0,04	0,17	0,09	0,24	0,35						
Value Added *at basic prices*	0,45	0,77	0,27	0,29	0,34	0,59						
Industry Output *at basic prices*	1,00	1,00	1,00	1,00	1,00	1,00						

Table 5-4: Identity matrix 6*6

1	0	0	0	0	0
0	1	0	0	0	0
0	0	1	0	0	0
0	0	0	1	0	0
0	0	0	0	1	0
0	0	0	0	0	1

Table 5-5: "I-A" matrix

0,875	0,000	-0,039	-0,002	-0,001	-0,001
0,000	0,977	-0,010	-0,218	-0,003	0,000
-0,152	-0,024	0,809	-0,026	-0,133	-0,039
-0,045	-0,004	-0,011	0,759	-0,001	-0,008
-0,006	-0,006	-0,003	-0,007	0,766	-0,022
-0,132	-0,088	-0,135	-0,077	-0,155	0,755

Table 5-6: Type I Leontief Inverse Matrix

	Agriculture, forestry and fishing	Mining	Manufactures	Utilities	Construction	Services
Agriculture, forestry and fishing	1,154	0,002	0,057	0,006	0,012	0,006
Mining	0,019	1,026	0,018	0,296	0,009	0,005
Manufactures	0,235	0,039	1,261	0,065	0,234	0,073
Utilities	0,075	0,008	0,025	1,322	0,010	0,016
Construction	0,017	0,012	0,012	0,021	1,316	0,039
Services	0,257	0,130	0,242	0,186	0,316	1,349

The **type II inverse matrix**[29] also shows the induced requirements (in terms of industry's output) of a production of one unit of a given industry's output. Its purpose is to take into account, besides the direct and indirect requirements included in the type I inverse matrix, the flows of money in and out of households and the effect of these flows on industries.

[29] This section builds on The Scottish Government, Statistics (2006).

The type II inverse matrix is derived in the same way as the type I inverse matrix. But since it is necessary to include households in the analysis we treat them as an additional industry by adding an extra row and column into the Direct Requirements matrix for "compensation of employees" and "final consumption expenditure by households" coefficients respectively.

The formal notation for this Direct Requirements matrix is:

A = $\begin{bmatrix} A_{II} & A_{IH} \\ A_{HI} & A_{HH} \end{bmatrix}$	Where	
	$(A_{II})_{ij}$	is the Direct Requirements matrix A, or the amount of industry i required per unit of industry j (considered above in the type I inverse matrix).
	$(A_{IH})_i$	is the amount of industry i required per unit of total household income <u>from all sources</u> (see note below).
	$(A_{HI})_j$	is the income paid to households per unit of output of industry i (compensation of employees divided by the total output of the industry).
	(A_{HH})	is the household expenditure per unit of exogenous household income. (This cell is set to zero).

Total household income from all sources is used as the denominator when calculating household expenditure coefficients (A_{IH}) even though it may at first seem odd not to use the total household expenditure figure from the I-O tables (in the example below called Total use at purchasers' prices). However, the total figure of household expenditure from the I-O tables includes household purchases that are bought with unearned income (pensions, dividends, etc). In other words, not all household expenditure results from 'Income from employment' paid to households. If the Total use at purchasers' prices figure were used as the denominator, the sum of A_{IH} would equal 1 and the resulting type II Leontief would tend to overestimate the induced effects of changes in the economy by artificially inflating the effect of earned income in generating further rounds of household spending.

Example, *cont.*
Centraal Bureau voor de Statistiek (2001) gives a figure of total household income for The Nederlands from all sources of 210485 million Euros 2000; we use this figure as the denominator when calculating household expenditure coefficients (AIH). Coefficients for the household sector can now be included in the Direct Requirements matrix (Table 5-7).
Following the same procedure as described above: subtracting matrix A from an identity matrix of the same dimensions (7*7) and calculating the inverse of the result (L=(I-A)-1) yields the type II Leontief inverse (Table 5-8).

Table 5-7: Direct Requirements matrix

	Agriculture, forestry and fishing	Mining	Manufactures	Utilities	Construction	Services	Final consumption expenditure by households
Agriculture, forestry and fishing	0,12	0,00	0,04	0,00	0,00	0,00	0,00
Mining	0,00	0,02	0,01	0,22	0,00	0,00	0,00
Manufactures	0,15	0,02	0,19	0,03	0,13	0,04	0,08
Utilities	0,04	0,00	0,01	0,24	0,00	0,01	0,03
Construction	0,01	0,01	0,00	0,01	0,23	0,02	0,00
Services	0,13	0,09	0,13	0,08	0,16	0,25	0,59
Compensation of employees	0,11	0,04	0,17	0,09	0,24	0,35	0,00

Table 5-8: Type II Leontief Inverse Matrix

	Agriculture, forestry and fishing	Mining	Manufactures	Utilities	Construction	Services	Final consumption expenditure by households
Agriculture, forestry and fishing	1,159	0,004	0,063	0,010	0,022	0,015	0,020
Mining	0,024	1,028	0,024	0,300	0,018	0,014	0,019
Manufactures	0,292	0,060	1,328	0,111	0,335	0,180	0,218
Utilities	0,094	0,015	0,048	1,338	0,045	0,052	0,074
Construction	0,028	0,015	0,024	0,029	1,334	0,059	0,040
Services	0,575	0,249	0,611	0,442	0,877	1,945	1,206
Compensation of employees	0,389	0,146	0,452	0,312	0,686	0,728	1,475

Once the type I (or type II) inverse matrices have been derived, the calculation of multiplier and effects is quite basic.

The multipliers allow users to **make estimates of the effects of changes in the economy**. For instance, if there is **an increase in final demand** (defined as a change in sales to the final consumers of goods and services for a particular good), we can assume that there will be an increase in the output of that commodity, as producers react to meet the increased demand; this is the **direct impact**. As these producers increase their output, there will also be an increase in demand on their suppliers and so on down the supply chain; this is the **indirect impact**. These two cumulative types of impacts can be calculated using the **type I** inverse matrix.

As a result of the direct and indirect impacts, the level of household income throughout the economy will increase as a result of increased employment; a proportion of this increased income will be re-spent on final goods and services: this is called the **induced effect**. This effect, along with the direct and indirect impact, is taken into account when the **type II** inverse matrix is used for the calculation of the multipliers.

Five different types of aggregate multipliers for both type I and type II effects can be calculated. (Those multipliers are derived for our example in Table 5-9 and Table 5-10).

5.2.1 Output multiplier $(O_{MULT})_j = \Sigma_i L_{ij}$

The Type I output multiplier for a particular industry is defined as the total of all outputs from each domestic industry required in order to produce one additional unit of output: that is, the column sums (Σ_i) from the Type I Leontief inverse matrix (L_{ij}). Similarly, the Type II output multiplier is given from the column sums of Industry rows (i.e. exclude compensation of employees) from the Type II Leontief.

Multiplying a change in final demand (direct impact) for an individual industry's output by that industry's type I (respectively type II) output multiplier will generate an estimate of direct + indirect (respectively direct + indirect + induced) impacts on output throughout the economy.

5.2.2 Income multiplier (IMULT)j = Σi viLij / vj

The income multipliers show the increase in income from employment (IfE) -or compensation of employees- that result from a change of €1 of income from employment in each industry. In the formula above, ‘v’ refers to the ratio of IfE/total output for each industry (last row in the augmented Direct Requirements matrix).

The multipliers show the ratio of direct plus indirect (plus induced if type II multipliers are used) income changes to the direct income change.

5.2.3 Income effects (Ieff)j = Σi viLij

This statistic shows the impact upon income from employment (IfE) -or compensation of employees- throughout the studied economy arising from a unit increase in final demand for industry j’s output.

While direct and indirect impacts are calculated using type I multipliers, type II multipliers also include induced effects in the economy.

If data on the level of employment (in FTE, full-time equivalent) is available for each of the industry sectors[30] employment effects and multipliers can be calculated as well.

5.2.4 Employment multiplier (EMULT)j = Σi wiLij / wj

The employment multipliers show the total increases in employment throughout the economy resulting from an increase in final demand which is enough to create one additional FTE employment in that industry. In the formula above, ‘w’ is equal to FTE per € of total output for each industry.

The multiplier is the ratio of direct plus indirect (plus induced if Type II multipliers are used) employment changes to the direct employment change.

[30] This is unfortunately not the case in our example.

5.2.5 Employment effects $(Eeff)j = \Sigma i\ wiLij$

The employment effects statistic calculates the impact upon employment throughout the economy (direct and indirect effect if type I inverse matrix is used, augmented by the induced effect if type II inverse is used) arising from a change in final demand for industry j's output of 1 unit.

Table 5-9: Type I, output and income multipliers

	Output multiplier	Income multiplier	Income effects
Agriculture, forestry and fishing	1,757	2,466	0,263
Mining	1,216	2,333	0,099
Manufactures	1,615	1,840	0,306
Utilities	1,896	2,372	0,212
Construction	1,898	1,953	0,465
Services	1,487	1,417	0,493

Table 5-10: Type II, output and income multipliers

	Output multiplier	Income multiplier	Income effects
Agriculture, forestry and fishing	2,173	3,6377	0,3887
Mining	1,372	3,4423	0,1456
Manufactures	2,098	2,7144	0,4515
Utilities	2,230	3,4996	0,3124
Construction	2,631	2,8813	0,6858
Services	2,265	2,0907	0,7279

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6 On the use of discrete choice models for modelling non-market behaviour in SSAs

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6.1 Introduction

The purpose of this report is to briefly introduce discrete choice models (sections 6.2 and 6.3) and to illustrate how such models can be used in the SSAs by two examples from SSA Himmerfjärden (sections 6.4 and 6.5).

6.2 What are discrete choice models?

Dependent variables in models are often discrete rather than continuous, which implies that there are many cases where conventional regression analysis is not suitable to apply. By "discrete dependent variables" we refer to cases when the dependent variable takes values 0,1,2,… Such values are sometimes meaningful in themselves, for example, when a dependent variable *y* is the number of persons in a family. But most often the values 0,1,2,… are instead codes for some qualitative outcome. Greene (1997, p. 872) gives the following examples:

- "**Labor force participation:** We equate "no" with 0 and "yes" with 1. These are qualitative choices. The zero/one coding is a mere convenience.
- **Opinions of a certain type of legislation:** Let 0 represent "strongly opposed", 1 "opposed", 2 "neutral", 3 "support" and 4 "strongly support". These are **rankings**, and the values chosen are not quantitative but merely an ordering. The difference between the outcomes represented by 1 and 0 is not necessarily the same as that between 2 and 1.
- **The occupational field chosen by an individual:** Let 0 be clerk, 1 engineer, 2 lawyer, 3 politician, and so on. These are merely categories, giving neither a ranking nor a count."

The typical approach to statistical analysis of models involving discrete dependent variables is similar to conventional regression analysis in the sense that these models try to relate the discrete outcome to a number of explanatory variables. This is done by applying various probability models where the probability that y takes a particular value j, i.e. $P(y=j)$, is viewed as a function of a vector of explanatory variables ($\boldsymbol{x}$) and their associated parameters (β), i.e. $P(y=j) = F(\beta'\mathbf{x})$. A specification of this function requires an assumption of some probability distribution such as the normal distribution and the logistic distribution.

6.3 The random utility model

The estimation of the discrete choice model might be made *ad hoc* by simply selecting a probability model that fits the data available. However, it could also be based on more explicit behavioural assumptions such as the random utility model (RUM). For example, a RUM setting is often a point of departure for environmental valuation methods such as the travel cost method and various stated preferences methods including the contingent valuation method and choice experiments (e.g., Haab and McConnell, 2002, Hensher et al., 2005).

In a RUM, an individual is viewed as choosing between J alternatives, which is described by a vector of attributes ($\boldsymbol{a}$). This means that the indirect utility of alternative i for individual k can be written as $v_{ik} = V_{ik}(\boldsymbol{a}_i, M_k - p_i)$, where M_k is the income of individual k and p_i is the cost incurred when selecting the ith alternative. Given that the individual is characterized by a utility maximizing behaviour, alternative i is chosen if and only if:

$V_{ik}(\boldsymbol{a}_i, M_k - p_i) > V_{jk}(\boldsymbol{a}_j, M_k - p_j)$ for all $j \neq i$

An individual is assumed to know her preferences and to maximize her utility in every choice made. However, these preferences are not known by the researcher, for whom utility therefore appears to be a random variable. An error variable (ε) is included in the utility function in order to capture this randomness, which means that the condition above can be written as:

$V_{ik}(\boldsymbol{a}_i, M_k - p_i, \varepsilon_{ik}) > V_{jk}(\boldsymbol{a}_j, M_k - p_j, \varepsilon_{jk})$ for all $j \neq i$

The introduction of randomness implies that it is now adequate to express the condition in terms of the probability that individual k chooses alternative i:

$P_{ik} = P(V_{ik}(\boldsymbol{a}_i, M_k - p_i, \varepsilon_{ik}) > V_{jk}(\boldsymbol{a}_j, M_k - p_j, \varepsilon_{jk}); \forall\, j \neq i)$

An empirical version of this RUM model requires a specification of the probability distribution of the error term and the functional form of the utility function. Some common assumptions are the following:

1. ε is entered into the utility function as an additive term
2. ε has an extreme value type I distribution
3. the utility function is a linear function of the attributes, e.g. $v_{ik} = \beta_1 a_{1i} + \beta_2 a_{2i} + \beta_M (M_k - p_i)$ in a case with two attributes and $M_k - p_i$ as a third explanatory variable

These assumptions constitute the basis for the conditional logit model, i.e. the probability that individual k chooses alternative i can be computed as:

$$P_{ik} = \frac{\exp(\beta_1 a_{1i} + \beta_2 a_{2i} + \beta_M (M_k - p_i))}{\sum_{j=1}^{J} \exp(\beta_1 a_{1j} + \beta_2 a_{2j} + \beta_M (M_k - p_j))}$$

where the parameters can be estimated through applying standard statistical software packages. However, some packages such as LIMDEP and NLOGIT (see http://www.limdep.com), include particularly many pre-defined estimation procedures for various types of discrete choice models, i.e. there is no need for the users to specify own likelihood functions even for quite advanced and complicated models.

6.4 A travel cost study applied to SSA Himmerfjärden

The EXTEND model for SSA Himmerfjärden considers, for example, the results of various policy options related to reductions of nutrient loadings to the sea. One probable result is an increased Secchi depth. The benefits of such an increase are obtained from applying an earlier travel cost study of the Stockholm archipelago, of which SSA Himmerfjärden is a part. Using a RUM setting and a conditional logit model, Soutukorva (2005) estimated the value of a one-metre Secchi depth improvement in the Stockholm archipelago to 9-29 million EUR (85-273 million SEK) per year (1 EUR = 9.4 SEK). This study was based on a mail questionnaire survey sent to a random sample of residents in the two counties of Stockholm and Uppsala. The vector of attributes ***a*** consisted of three variables considered to explain the respondents' choices of recreational sites in the archipelago: (i) the cost of travelling to the sites including the opportunity cost of travel time, (ii) the bathing water quality at sites as measured by Secchi depth, and (iii) the accessibility to sites by public ferry.

A common problem in travel cost studies is the presence of multi-purpose trips, i.e. respondents have more than one purpose when visiting a recreational site, such as both bathing and visiting a restaurant. Soutukorva (2005) approached this problem by letting the respondents in the survey mark the importance of water clarity for their site choice on a continuous scale. For respondents who put a mark on the right end of the scale ("vital importance"), 100 per cent of the travel cost was included in the estimation. When water clarity was of less importance, travel costs were adjusted accordingly. For those respondents who stated that water clarity was of no importance for their choice of site, travel costs were set to zero in the estimation.

Using the part of the survey data that concerned SSA Himmerfjärden, Kinell (2008) also estimated a conditional logit model, which gave the results reported in Table 6-1. Model A and B refer to a specification excluding and including the accessibility by public ferry variable, respectively. c is the intercept, and β_{tctime}, β_{sd} and β_{ferry} refer to the parameters associated with the three explanatory variables of travel cost, Secchi depth and accessibility by public ferry.

The estimates in Table 6-1 are the basis for calculating the compensating variation as a monetary measure of the change in human wellbeing due to a Secchi depth improvement in SSA Himmerfjärden. Compensating variation is a measure of the change in the (Hicksian) consumer surplus, and as is explained in more detail in the SPICOSA-WP2 deliverable *D.2.1 Economic Assessment*, an individual's consumer surplus is equal to the difference between the maximum amount of money that he/she is willing to pay for consuming a particular amount of a good and what he/she actually has to pay.

The change in consumer surplus is therefore used in economics as a measure of change in wellbeing. See also, e.g., Freeman (2003).

Table 6-1: Estimated coefficients (p-values in parentheses)

	Model A	*Model B*
c	-4.539590 (0.00)	-4.506779 (0.00)
β_{tctime}	-0.000960 (0.01)	-0.002184 (0.00)
β_{sd}	0.078781 (0.00)	0.056435 (0.00)
β_{ferry}		0.079149 (0.00)
LR statistics	56.9 (0.00) 2df	245.02 (0.00) 3df

Compensating variation for a changed Secchi depth is obtained as (see, e.g., Haab and McConnell, 2002, p.224):

$$CV = \frac{\ln\left\{\sum e^{\left(v_i^1\right)}\right\} - \ln\left\{\sum e^{\left(v_i^0\right)}\right\}}{\gamma}$$

where superscript 0 (1) denotes the initial (final) Secchi depth level and γ is the marginal utility of income. In the case of an increase in water clarity, compensating variation is the maximum willingness to pay for obtaining such an improvement. For example, computing compensating variation for the particular case of a one-metre Secchi depth improvement in Himmerfjärden results in the estimates presented in Table 6-2 below (1 EUR= 9.4 SEK). This is an example of the results that also will be produced in the EXTEND model.

Table 6-2: Aggregate CV per year for a one-metre secchi depth improvement in Himmerfjärden

Explanatory variables included in the model	**CV, EUR/year (SEK/year)**
A: Secchi depth, travel cost (including value of time)	170 151 (1 599 420)
B: Secchi depth, public ferry and travel cost (including value of time)	33 784 (317 566)

While the compensating variation estimate is of great interest because it can be included in an economic evaluation (through cost-benefit analysis) of various policy options for reducing the nutrient load to SSA Himmerfjärden, the logit model can also produce other useful results. For example, since the model relates the probability of selecting a site to a number of explanatory variables, it can also predict how a change in an explanatory variable affects this probability. This means that the estimated model can be used for saying something about how a change in Secchi depth is likely to affect the number of visitors to SSA Himmerfjärden.

This issue was approached by estimating a quality elasticity of demand or, more precisely, the following elasticity of the probability of a visit to Himmerfjärden as the Secchi depth improves (see Ben-Akiva, 1994, or equation (24) in Kinell, 2008, for further explanations):

$$E_{a_{sd,i}}^{P_i} = \frac{\partial \ln P_i}{\partial \ln a_{sd,i}} = [1 - P_i] a_{sd,i} \beta_{sd}$$

This elasticity of the probability of a visit to Himmerfjärden as the Secchi depth improves was computed as a mean of the elasticities estimated for the recreational sites belonging to Himmerfjärden SSA. The elasticity indicates a positive relationship between Secchi depth improvement and number of visits to Himmerfjärden.

The next step is to compute the probability of a visit to Himmerfjärden. This probability is estimated by computing the number of visits to Himmerfjärden as a share of the total number of visits to the whole of Stockholm archipelago. This gives a probability of about 0.06, which corresponds to about 231 000 visits[31] per year to Himmerfjärden. Recall that all estimations are based on results from the survey.

The estimated elasticity was subsequently used for computing the increase in the number of visits to Himmerfjärden because of a small (0.1-metre) Secchi depth improvement; see Table 6-3 for results for the models A and B. The additional number of visits was calculated by multiplying the annual number of visits to Himmerfjärden (about 231 000) by the increase in the probability of a visit to Himmerfjärden after a 0.1-metre Secchi depth improvement.

Table 6-3: Change in the number of visits to Himmerfjärden following a 0.1-metre Secchi depth improvement

Model	Number of additional visits
A	3040
B	4180

Note: The calculations are based on the coefficients estimated in the models (A-B).

The fact that a Secchi depth improvement tends to result in more people visiting Himmerfjärden introduces a feedback loop in the EXTEND model because it influences aggregate compensating variation. In the SSA work in the beginning of 2009, we aim at refining this feedback loop and also consider

[31] Note that this number of visits constitutes a lower boundary of the actual number of visits, because the travel cost study only collected data on visits actually involving a travel to Himmerfjärden. For example, visits to Himmerfjärden that take place by simply walking from one's summer house to a beach are not included.

how to introduce a non-linear – and thus more realistic – relationship between compensating variation and Secchi depth. Dynamic features in the EXTEND model might also be introduced by studying, for example, the influence of increases in income and population. Such increases would affect the opportunity cost of travel time (and thus compensating variation) and aggregate compensating variation, respectively.

6.5 A choice experiment for social evaluation of one of the policy options in SSA Himmerfjärden

In environmental economics, choice experiments are typically applied for obtaining information on people's willingness to pay for individual environmental attributes. This information is inferred from surveys where respondents are asked to make repeated choices among different levels of the environmental attributes and a cost attribute, where the latter is presented as the cost for obtaining the levels of environmental attributes. However, a choice experiment approach might be useful also in other settings when the relative importance of various attributes for people's choice behaviour is to be analyzed.

In SPICOSA, policy options identified in the SSAs are to be evaluated from an environmental, economic and social point of view. The social evaluation might include, for example, to what extent a policy option is acceptable among the stakeholders that would be involved in carrying out the option and/or influenced by the option. This evaluation is likely to be partly qualitative and partly quantitative, where the latter can be a part of the EXTEND model of SSA Himmerfjärden. As a basis for the quantitative evaluation, we aim at applying a choice experiment approach to one particular policy option: restoring and constructing wetlands in the agricultural landscape as a measure for reducing the nutrient loadings to the sea.

The degree to which this policy option can be carried out depends on the willingness among farmers to convert a part of their land to wetlands. It has earlier been illustrated that this willingness is not only dependent on economic incentives, but also on other aspects related to how authorities design a wetland and riparian zone program (Söderqvist, 2003). This suggests the design of a choice experiment aiming at the estimation of what we call a participation function, i.e. a function relating the probability that a farmer chooses to participate in a (hypothetical) program to a number of attributes describing the program.

In SSA Himmerfjärden, exploratory face-to-face interviews with farmers have indicated the relevant attributes of such a program. Given this information, a mail questionnaire has been developed and a pilot study is at present being carried out among a small random sample of farmers in the catchment area of SSA Himmerfjärden. The main survey is planned to be executed in the beginning of 2009. The program attributes selected for the pilot study were the following:

1. Compensation for construction costs - the extent to which the authorities will cover the costs that farmers incur in construction of a wetland (in percent);
2. The maximum grant available for wetland construction (in SEK);
3. The annual grant for maintenance of constructed wetlands (in SEK);
4. The time horizon for farmers' obligations to maintain the wetland (in years);
5. The existence of a particular wetland project/forum in which farmers and other stakeholder groups are represented (yes or no).

Firstly, the current settings for all attributes are presented for the respondent. Then, all attributes will be binary and varied in each choice set, i.e.; either an attribute is at the current level or it is changed to another level – our hypothesis is that such a change will tend to increase farmers' willingness to participate. For each choice set, different attribute levels are changed, which forces the respondent to make trade-offs between different attributes. If the choice sets are well formulated and designed, the study will provide information of to what extent different attributes are important for farmers' willingness to participate in wetland construction. This information could indicate how to develop and design wetland projects and policies/compensation levels for wetland construction in the future.

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