VALUING BIODIVERSITY BENEFITS FOR VULNERABLE GOURPS

BY

PAULO A.L.D. NUNES, HELEN DING AND ANDREA GHERMANDI

Introduction

This paper proposes the development and empirical application of economic valuation methodologies to assess the socio-economic magnitude of biodiversity benefits and their contribution to the support of human livelihoods, in particular rural poor communities. The present analysis will embrace an economic valuation perspective that (1) quantifies the degree of *dependency* of the local economies with respect to biodiversity and ecosystem services, including their role in the creation of employment/income opportunities to the communities, and (2) assesses the degree of *vulnerability* of the local economies to the changes of provisioning of biodiversity and ecosystem services, such as regulation of climate, soil and water in the environment.

The economic valuation will be applied firstly within the European context, with a potential to be extended to a global scale. Moreover, we will provide a number of case studies to demonstrate how economic valuation of ecosystem services can be conducted as well as to quantify the dependency and vulnerability of human livelihoods associated with biodiversity and ecosystem services at country level. Finally, the present study focuses on a set of main biodiversity benefits and ecosystem services among the most relevant ecosystems, which provide substantial environmental (cash and non-cash) incomes to humans – see Table1. Particular attention is paid to quantifying the biodiversity benefits within vulnerable communities, by exploring the United Nations millennium development goals to identify and profile the vulnerable groups in various classified world economies.

Table 1: Ecosystem services' values reported

		Ecosystems			
		Forest	Marine coastal	Wetland and freshwater	Agriculture
Evaluated Ecosystem Services	Provisioning Service	●	○	●	○
	Regulating Service	●	○	●	○
	Cultural Service	●	●	●	○

Note: ● - Economic valuation is completed; ○ - Economic valuation is in progress.

1. SETTING THE SCENE: LINKAGES BETWEEN BIODIVERSITY, ECOSYSTEM SERVICES AND HUMAN LIVELIHOODS

> ***"… More than a billion people now live within the world's 19% forest biodiversity 'hotspots' and population growth in the world's tropical wilderness area is 3.1 percent, over twice the world's average rate of growth. Over 90 percent of those who live on less than a dollar a day depend fully or in part on forest products for their livelihoods."*** **Scherr et al. (2003)**

1.1 Introduction

Biodiversity is a multi-dimensional concept, which is defined as the quantity and variability among living organisms within species (genetic diversity) between species and between ecosystems, underpinning the supply of a variety of ecosystem services from which humans can benefit directly or indirectly. However, the contribution of biodiversity to human welfare and human livelihoods reveals to have a complex nature. Recently, the United Nations' Millennium Ecosystem Assessment (MEA) proposed to study this relationship exploring the use of ecosystems services, which are interpreted by the socio-economic scientists as biodiversity benefits (MEA, 2005). Therefore, the contribution of biodiversity to human wellbeing can be examined by the provision of ecosystem services. The MEA classification states that all ecosystem services can fall into four major categories, i.e. provisioning, regulating, cultural and supporting services, which directly or indirectly influence the human well-being – see Figure 2.1.

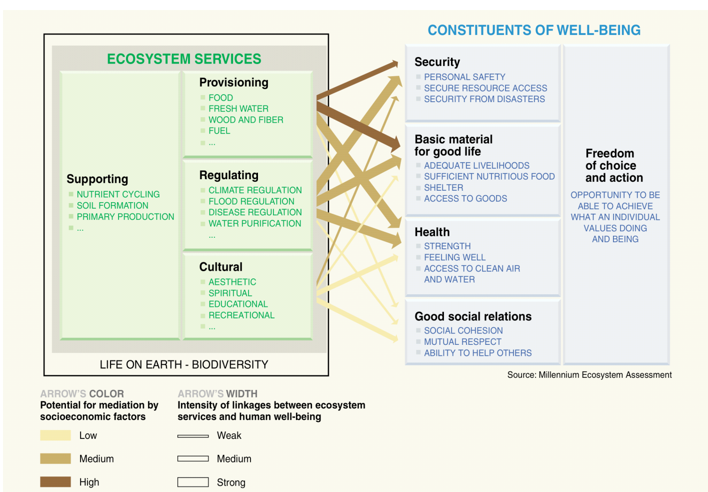

Figure 2.1: Linkages between ecosystem services and human well-being (Source: MEA, 2005, pp. iv)

Therefore, the implications of biodiversity to the support of human livelihoods, including the ones of vulnerable groups such as the rural poor, can be examined by the intensity of the linkage between ecosystems, and services provided (also known as biodiversity benefits), and the constituents of human wellbeing. This includes the examination of ecosystem services such as the provision of food and water, disease management, climate regulation, flood control, spiritual fulfillment, and aesthetic enjoyment. These have been recognized as having an essential role in achieving the United Nation's Millennium Development Goals (UNEP-WCMC, 2007). In the present report, we shall embrace a conceptual model for valuing the linkages of biodiversity benefits on human livelihoods that sheds light on two distinct value transmission mechanisms. The first captures the market value components of the biodiversity benefits on human livelihoods. A second component encapsulates the non-market aspects dimensions. This approach will be presented and discussed in the following sub-sections.

1.2 Conceptual model for mapping the linkages of biodiversity benefits and human livelihoods

Human livelihoods are defined as comprising the capabilities, assets (including natural capital, physical capital, human capital, financial capital and social capital) and activities, such as crops, livestock, extractive activities, wage employment and own business, that are required for a means of living (Babulo et al., 2008). This definition indicates that biodiversity benefits and ecosystem services substantially contribute to people's livelihoods in terms of direct increase in people's revenues as well as the enhancement of non-income benefits from the received ecosystem services. In other words, we can argue that biodiversity benefits, and ecosystem services, are linked with human livelihoods in two ways – see Figure 2.2.

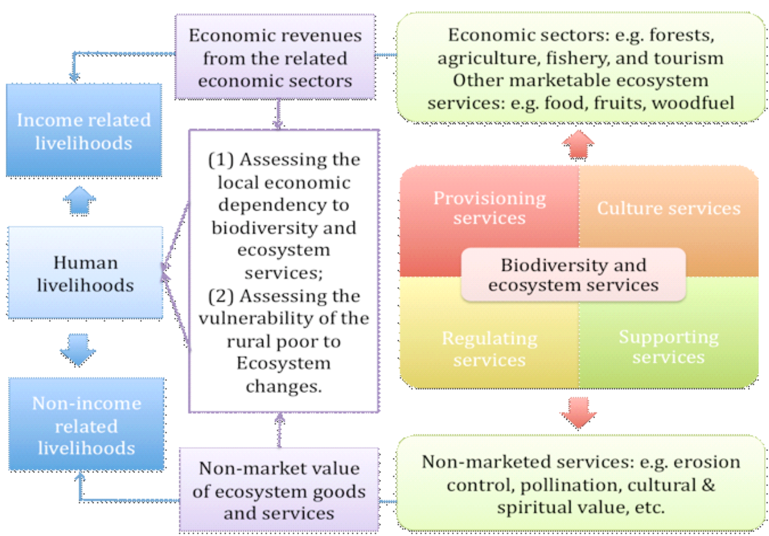

Figure 2.2: Framework of assessing the human livelihoods through biodiversity and ecosystem services

On the one hand, ecosystem services are essential inputs for many economic sectors, including forestry, agriculture, fishery, and tourism or direct source of income/revenues to the local communities (consumers/firms) who are involved in markets trading ecosystem services, such as food, fruits and wood fuel, among others. The strength of this linkage can be estimated through a systematic economic sector analysis, and the results reflect the degree of *dependency* of the local economies with respect to biodiversity and ecosystem services, including their role in the creation of employment/income opportunities to the communities. In this context, valuing the economic revenues that rural dwellers or the poor local communities can extract from the use of environmental resources enables us to assess its quantitative contribution to rural livelihoods and the extent of dependency of rural people on nature products and ecosystem services.

On the other hand, ecosystem services also contribute to non-income related livelihoods. The ecosystem regulating and supporting services will safeguard the living environment as well as guarantee the continuous economic activities of humans, in particular the rural dwellers; whereas the ecosystem cultural services are essential to the spiritual and cultural value of the local communities. The strength of this linkage can be estimated through a systematic economic analysis of the non-income related value of biodiversity and ecosystem services on human livelihood systems, which in turn will allow us to complement the understanding of the degree of *dependency* of the local economies with respect to biodiversity and ecosystem services. Moreover, both value transmission mechanisms will allow us to understand the degree of *vulnerability* of the local economies, in particular rural poor, with respect to changes, or losses, of biodiversity and the respective impacts in the provision of ecosystem services.

1.3 An hybrid economic model for valuing the magnitudes of biodiversity benefits on human livelihoods

As previously shown, ecosystem provides an array of services to human wellbeing in terms of provisioning services, cultural services, regulating services and supporting services (MA, 2005), many of which are associated with a variety of economic activities that involve either direct or indirect employment in the related economic sectors and lead to direct changes of incomes to the relevant households; some others will contribute to direct welfare enhancement of the local communities derived from their enjoyment of nature resources, knowledge of biodiversity and ecosystem as well as other non-marketed values of biodiversity and ecosystems. Due to the complex nature of biodiversity and ecosystem services, economic valuation of the biodiversity benefits is not always straightforward. Different economic valuation approaches are needed. In fact, economists have been applying different valuation methodologies to estimate the economic values of ecosystems and the services they provide.

In Table 2.2, we summarize the standard ecosystem valuation techniques that are mostly used in the literature – see Appendix I for more detailed explanation on each methodology. It is important to note that in the economic theory, socio-economic valuation of biodiversity and ecosystems is anchored in the assessment of changes in the productivity of the economic sectors under concern and/or respective consumer’s utility and requires the investigation of appropriate microeconomic valuation techniques, including both market-based economic valuation tools (e.g. market price analysis) and non-market valuation tools (e.g. contingent valuation methods, travel costs methods, meta-analysis, and value transfer methods). Therefore, the estimated economic benefits of biodiversity and ecosystems should reflect the welfare changes of the individuals being directly affected by changes in biodiversity and ecosystems, or the average welfare change of the individuals in a considered population (Nunes et al. 2003).

Table 2.2: Tool box of economic valuation techniques

Category	Technique	Description	Example
Revealed preference approaches			
Market prices	Market prices	How much it costs to buy an ecosystem good or service.	The price of timber or mineral.
Production function approach	Effect on production	Relates changes in the output of a marketed good or service due to the changes in a measurable amount of production inputs.	The reduction in lifespan of a hydro.
Surrogate market approach	Travel costs	Valuation method based on the willingness to pay for recreational/ leisure use of nature resources, derived from the amount of time and money people spend on visiting a relevant ecosystem.	The transport and accommodation costs, entry fees and time spent to visit a natural park.
	Hedonic pricing	The difference in property prices or wage rates that can be ascribed to the different ecosystem quality or values.	The difference in house prices between those overlooking an area of natural beauty and those without a view of the landscape.
Cost-based approach			
Damage costs avoided		The costs incurred to property, infrastructure, production when ecosystem services which protect economically valuable assets are lost, in terms of expenditure saved.	The damage to roads, bridges, farms and property resulting from increased flooding after the loss of catchment protection forest.
Stated preference approaches			
Contingent valuation method		Infer ecosystem value by asking people directly what is their willingness to pay (WTP) for resource conservation or willingness to accept for (WTA) compensation for the loss of biodiversity/ ecosystems	How much would you be willing to contribute towards a fund to clean up and conserve a river?
Conjoint analysis		Elicit information on preferences between scenarios involving ecosystems between which the respondents would have to make a choice, at different price or cost saved.	The relative value wildlife, landscape and water quality attributes of a river under different conservation scenarios, relative to the status quo.
Choice experiments		Presents a series of alternative resource or ecosystem use options, each defined by various attributes including price and asks respondents to evaluate these "sets", which each contain different bundles of ecosystem services.	Respondents' preferences for conservation, recreational facilities, and educational attributes of natural woodlands.
Value transfer approaches			
Meta-analysis		This technique takes the result from a number of studies and analyses in such a way that the variation in value of ecosystem services obtained in those primary studies can be explained.	Analysis of many primary contingent valuation studies for woodlands to derive the trends in the key variables affecting visitor WTP values for woodlands, to establish a suitable variable for adjustments for the site to be assessed.

Source: adapted from WBCSD (2009)

The monetary valuation exercise of the present report is based on the application of an integrated, hybrid valuation model. It is hybrid, because the model is characterized by the use of both of biophysical and economic valuation models and *integrated* because it is characterized by an integrated use of alternative economic valuation methodologies. The combination of the use of valuation techniques for various ecosystem goods and services are presented in Figure 2.3, including market price analysis methods, cost assessments methods and meta-analysis-based value transfer methods. These techniques are most appropriately applied in the context of regional or national scale ecosystem changes, disaggregated by sector or market. The use of the techniques in isolation (sometimes referred to as 'bottom-up studies) is predicated on an assumption that any incremental damage in ecosystems will not have large, indirect (non-marginal) impacts, affecting the prices of a range of goods and services that flow through the macro-economy.

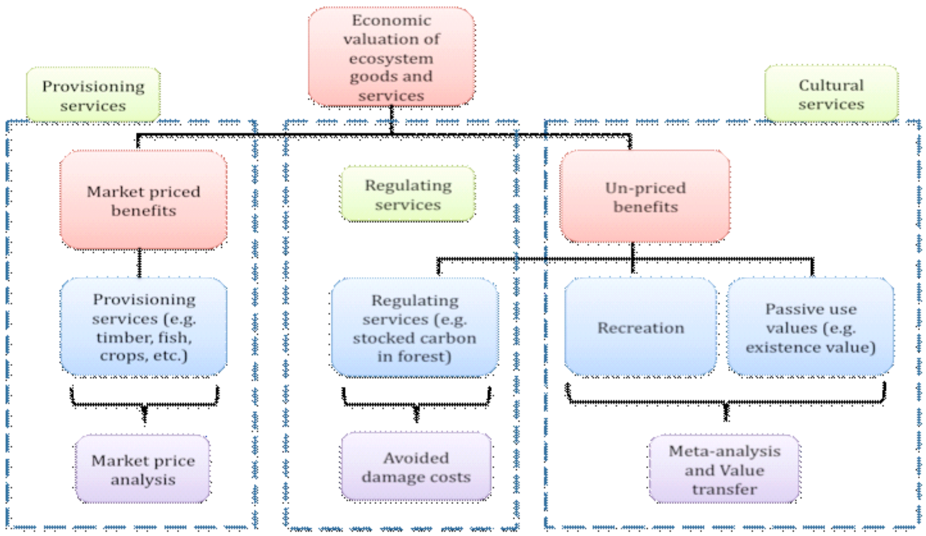

Figure 2.3: A hybrid economic valuation methodology *(adapted from Ding et al. 2010)*

Moreover, Figure 2.3 also shows that benefits derived from ecosystem provisioning services are estimated in terms of direct financial returns from the related economic sectors that using ecosystem goods and services as production inputs. Since market information is readily available for this kind of services, a direct market pricing method is used in the valuation exercise. The estimated financial revenues are used as a proxy of annual income of the local population/households, whose livelihoods are directly/indirectly influenced by the ecosystem. As regards the regulating services, such as carbon cycle regulation, there is no market available for trading the service, therefore damage costs avoided method is applied to value the expenditure saved (thus the benefits gained) from not losing economically valuable assets due to the loss of ecosystem services under social and environmental drivers. Finally, the value of cultural services, which consist of recreational value and passive–use value of ecosystems, is captured by using revealed preference (e.g. Travel Cost Methods and Hedonic Pricing) or stated preference methods (e.g. Contingent Valuation Method and Choice Experiments) in a surrogate market. However, given the limited resources available for the present study, no original valuation studies using revealed and stated preference methods are conducted. Instead, a meta-analysis based value transfer method is used in to estimate the WTP for receiving or increasing cultural services provided by ecosystems.

We shall present the empirical application of the hybrid valuation model to three ecosystems, including forest, wetlands/freshwater and marine/coastal systems in 32 European countries, including (1) Mediterranean Europe - Greece, Italy, Portugal, Spain, Albania, Bosnia and Herzegovina, Bulgaria, Serbia and Montenegro, Turkey; (2) Central and Northern Europe - Austria, Belgium, Denmark, Estonia, France, Germany, Ireland, Latvia, Lithuania, Luxembourg, Netherlands, Switzerland, Croatia, Czech Republic, Hungary, Poland, Romania, Slovakia, Slovenia, United Kingdom; (3) Scandinavian Europe - Finland, Norway, Sweden. This way we shall be able to evaluate the potential of these in terms of respective (market and non-market) support to human livelihoods. This will be the core of section 10. Before we shall proceed with the study of biodiversity and vulnerable groups in Europe, core of Section 9. In order to map biodiversity at European level, we explore the use of publically available information, coming from all various sources, at both country level and georeferenced maps. In the literature, a variety of indicators have been developed for evaluating biodiversity (e.g. the mean species abundance (MSA), species richness and the IUCN red-list index, etc) and socio-economics (e.g. GDP per capita, income poverty indicators, and population, etc). Ultimately, we will map the selected indicators in a spatial gradient, allowing us to identify the strength between biodiversity and human livelihoods.

The information collected in Section 9 and Section 10 will be used for producing GIS maps, which allows us to visualize strength of magnitudes with respect to the linkages between biodiversity, value of ecosystem services values, and vulnerable groups, and to discuss spatial coincidence of the three components at different geographical scales.

2. PROFILING BIODIVERSITY AND VULNERABLE GROUPS IN EUROPE USING INDICATORS

2.1 Biodiversity Spatial Profile in Europe

In order to characterize the spatial distribution of biodiversity in Europe, we rely on the index of biodiversity described in Wendland et al. (2009). Such index builds upon the information on species ranges of mammals, birds and amphibians from global vector data (Baillie et al., 2004; BirdLife International, 2006; IUCN, 2006) and combines it in a single index by weighing species ranges by their threat status as defined by IUCN's Red List (IUCN website, 2007). The technical details on the weighing procedure and construction of the aggregated index are given in Wendland et al. (2009). The final index is presented in a 30 arc second grid (approximately 1 km at the equator) and is mapped globally. In Figure 2.4, we present the distribution of the biodiversity index within Europe.

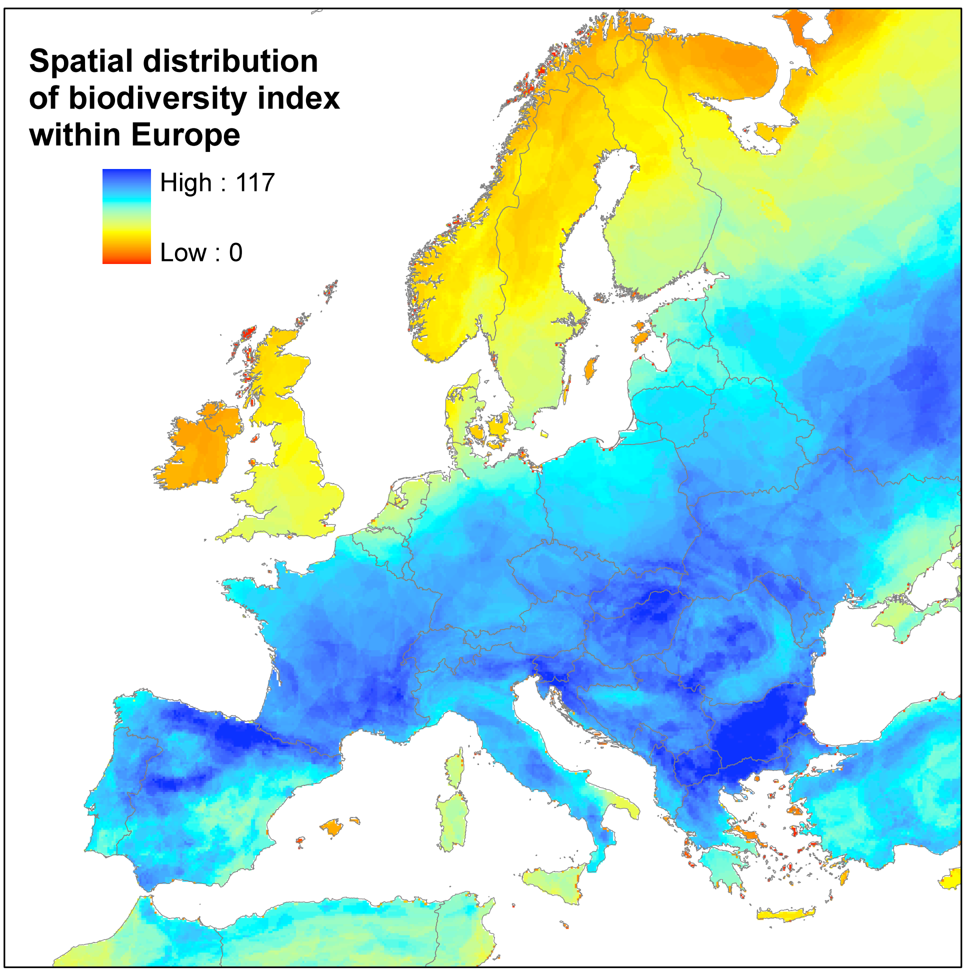

Figure 2.4: Distribution of terrestrial biodiversity within Europe *(based on Wendland et al., 2009)*

Figure 2.4 shows that terrestrial biodiversity is distributed unevenly in Europe. North European countries including Scandinavia, United Kingdom, and Ireland are characterized by relatively low biodiversity. The highest values of terrestrial biodiversity within Europe are found in East European countries, notably Bulgaria and Slovakia, and in the Northern regions of Spain. Within some countries there is an important range of variability in the index. For instance, in Italy high values of the biodiversity index are to be found in mountainous regions in the Alps and Apennines, while low-lying regions and, particularly, islands present lower values of the index. It is important to notice that, at a global scale, European countries score rather poorly in the biodiversity index compared to biodiversity hotspots in South America, Africa and South East Asia where the highest values of the index are found (up to 407).

Table 2.3: Biodiversity indicators for Europe

Country	Bird species (number)[1]	Mammal species (number)[1]	Reptile species (number)[1]	Vascular plant species (number)[1]	Biodiversity index[2]
Albania	303	73	37	3031	76.29
Austria	412	101	16	3100	76.52
Belgium	427	92	12	1550	62.13
Bulgaria	379	106	33	3572	91.31
Bosnia-Herzegovina	312	78	27	-	77.69
Switzerland	382	93	17	3030	77.35
Czech Republic	386	88	11	1900	76.60
Germany	487	126	15	2682	68.99
Denmark	427	81	8	1450	36.50
Spain	515	132	67	5050	70.81
Estonia	267	67	6	1630	54.85
Finland	421	80	5	1102	39.70
France	517	148	46	4630	76.46
United Kingdom	557	103	16	1623	34.73
Greece	412	118	63	4992	62.74
Croatia	365	96	34	4288	76.90
Hungary	367	88	18	2214	84.62
Ireland	408	63	6	950	22.93
Italy	478	132	55	5599	67.14
Lithuania	227	71	6	1796	67.32
Luxembourg	284	66	9	1246	71.94
Latvia	325	68	7	1153	60.33
Macedonia	291	89	29	3500	89.93
Netherlands	444	95	13	1221	49.16
Norway	442	83	7	1715	29.65
Poland	424	110	11	2450	70.82
Portugal	501	105	38	5050	68.75
Romania	365	101	22	3400	78.36
Serbia and Montenegro	381	96	35	4082	81.01
Slovakia	332	87	14	3124	83.67
Slovenia	350	87	29	3200	85.71
Sweden	457	85	7	1750	34.14

[1] Source: UNEP-WCMC (2004)
[2] Estimated by the authors based on the index in Wendland et al. (2009)

Table **2.3** 2.3 above summarizes the information on various biodiversity indicators assessed at country level. The data on the number of known bird, mammal, reptile, and vascular plant species was gathered from UNEP-WCMC (UNEP, 2004) and are compared to the average score of the biodiversity index by Wendland et al. (2009) for each European country. Overall, the highest biodiversity in terms of number of species of birds, mammals, amphibians, and vascular plants is in France, Italy, and Spain. All three countries are characterized by a relatively high value in the biodiversity index. It is reminded that the index is not constructed only based on the number of species but also on their threat status as defined by IUCN's Red List. Despite the smaller range of species, several Central and East European countries (e.g., Bulgaria, Macedonia, Slovenia) are characterized by a higher score in the biodiversity index. On the lower side of the range, countries such as Estonia, Ireland, Lithuania, Luxembourg and Latvia present the smallest range of animal and plant species. Ireland also has the lowest values among the considered countries for what concerns the biodiversity index. Notably, the United Kingdom is characterized by the largest number of known bird species but shows a relatively low diversity in reptiles and vascular plants and is characterized by a low value of the biodiversity index.

2.2 Identification of vulnerable groups in Europe using socio-economic poverty indicators

The selection of socio-economic poverty indicators is based on an extensive survey among the existing indicators in the literature, covering more than 34 different indicators within three major databases, including the UNDP-human development indicator, the World Bank - Environmental Economics and Human Wellbeing Indicator, and the UNSD - Millennium Development Indicators. From these, we focus on attention on 5 main categories of indicators. We refer to indicators on income and income inequality; demographic indicators; the World Bank’s analytical income categories; a set of poverty related indicators; and the value added of agriculture with respect to the GDP. All in all, we have about 10 different socio economic indicators are considered – see Table 2.4.

Table 2. 4:. A survey of socio-economic poverty indicators

Indicator	Type of Indicator	Data coverage	Type of data	Source of information
GDP per capita (US$)	Economy and inequality	194 countries	One year census data	UNDP-Human Development Indicator
Gini index	Economy and inequality	194 countries	One year census data	UNDP-Human Development Indicator
Urban population (% of the total)	Demographic trends	227 countries	Time series data	World Bank - Development Indicator
Population	Demographic trends	227 countries	Time series data	World Bank - Development Indicator
WB classification of world Economics	Wealth category	210 countries	Categorical data	World Bank - Development

				Indicator
Human poverty index (HPI-1) rank	Human and income poverty	164 countries	one year census data	UNDP-Human Development Indicator
Human Poverty Index (HPI-1) value (%)	Human and income poverty	164 countries	one year census data	UNDP-Human Development Indicator
Population living below 50% of median income (%)	Human and income poverty: OECD countries	30 countries	one year census data	UNDP-Human Development Indicator
Agriculture, value added (% of GDP)	Economy/ Forestry	227 countries	Time series data	World Bank - Development Indicator
Employment in forestry 1990 and 2000	Economy/ Forestry	229 countries	yearly average/one year	FAO - FRA 2005

Indicators of income and inequality. GDP per capita indicator is an important basic indicator showing economy and inequality of wealth distribution in different European countries. GINI index is another standard economic measure of income inequality, based on Lorenz Curve. A society that scores 0.0 on the Gini scale has perfect equality in income distribution and the score of 1.0 (or 100) indicates total inequality where only one person corners all the income. Therefore, higher is the number over 0, is higher the inequality. Demographic indicators such as population and "percentage of urban/rural population" provide important information for studying a country's rural/urban migration trend and per capita income in particular in the rural area.

The World Bank's analytical income categories (low, middle, high income) for classifying world's economies are based on the gross national income (GNI) per capita, which is calculated using the World Bank Atlas method. In Europe, however, all selected countries fall into upper middle and high-income categories with very low variation in the data.

Poverty related indicators are taken into consideration: (1) Human poverty index rank, (2) Human poverty index value, and (3) Population living below 50% of median income. Human poverty index is an indication of the standard of living in a country, developed by United Nations. The common international poverty line has been set $1.25 at 2005 Purchasing-Power Parity a day, which however cannot be applied to Europe, as all countries belong to higher income categories than developing countries. Therefore, we chose to use the indicator of population living below 50% of median income to describe the poverty level of the country.

Agriculture, value added (% of GDP) is the net outputs of agricultural sectors - incl. forestry, hunting, and fishing, as well as cultivation of crops and livestock production - after adding up all outputs and subtracting intermediate inputs. It is calculated without making deductions for depreciation of fabricated assets or depletion and degradation of natural resources.

For the reasons previously stated, three indicators are selected for the present analysis in Europe. The data are collected at national level and presented in Table 2.5.

Table 2.5: Selected socio-economic indicators for Europe

Country	GDP per capita (2007 $)	Urban (% of population) in 2007	Population living below 50% of median income
Albania	3,405	46	..
Austria	44,879	67	7.7
Belgium	42,609	97	8.1
Bosnia and Herzegovina	4,014	47	..
Bulgaria	5,163	71	..
Croatia	11,559	57	..
Czech Republic	16,934	74	4.9
Denmark	57,051	86	5.6
Estonia	15,578	69	12.4
Finland	46,261	63	6.5
France	41,970	77	7.3
Germany	40,324	74	8.4
Greece	27,995	61	14.3
Hungary	13,766	67	6.4
Ireland	59,324	61	16.2
Italy	35,396	68	12.8
Latvia	11,930	68	..
Lithuania	11,356	67	..
Luxembourg	103,042	83	8.8
Netherlands	46,750	81	4.9
Norway	82,480	77	7.1
Poland	11,072	61	11.5
Portugal	20,998	59	..
Romania	7,703	54	8.1
Serbia and Montenegro	5,620	56.5	..
Slovakia	13,891	56	7
Slovenia	23,379	49	8.2
Spain	32,017	77	14.2
Sweden	49,662	84	5.6
Switzerland	56,207	73	7.6
Turkey	8,877	68	..
United Kingdom	45,442	90	11.6

Source: World Bank - World development indicator; UNDP - Human development indicator

3. ESTIMATING THE ECONOMIC SIGNIFICANCE OF THE LINKAGES BETWEEN BIODIVERSITY, ECOSYSTEM SERVICES AND HUMAN LIVELIHOODS: CASE STUDIES IN EUROPE

3.1 Introduction

Based on data from 32 European countries, forests cover a surface of approximate 185 million ha (FAO, 2005), which accounts for about 32.7% of the territory. By classifying the forest areas in terms of their latitudes, it is easy to see that forest types are not uniformly distributed in Europe. For instance, in the Mediterranean Europe, most of the forests are coniferous and broadleaved evergreen forests, which account for 30% of the total forest area in the three regions. The Central and Northern European regions are home of most of the temperate forests, which account for 35% and 19% of the total forests, respectively. Finally, in the Scandinavian Europe forest area accounts for the remaining 16% of total forest, in which the identical forest biomes are mainly boreal. Due to the diverse climate conditions across latitudes, species diversity and dynamics of forest ecosystems differ considerably throughout Europe, as reflected in the numbers and composition of tree species. For instance, Ministerial Conference on the Protection of Forests in Europe MCPFE (2007) reported that about 70% of the forests in Europe are dominated by mixed forest consisting of two or several tree species, and the remaining 30% are dominated by one tree species alone, mainly by conifers. In addition to the natural conditions, the current European forest structure, in part, forest species composition has been heavily influenced by anthropophagic interventions, such as past land use and management (Ellenberg, 1986). In particular, driven by the forest protective management strategy in Europe, a 1.0 percent annual expending rate has been found in the area of mixed forests over the last 15-year period (MCPFE 2007), which partly may be because of the widely acknowledged scientific evidence that mixed forests being composed of several tree species are usually richer in biodiversity than the forests dominated by one tree species.

3.2 European Case Studies

3.2.1 Economic Valuation of Forest Ecosystems

1. Mapping the provision of ecosystem goods and services by European forests

A concise mapping of ecosystem goods and services (EGS) are basis for conducting high quality ecosystem assessment studies. For this reason, we adopt the MA approach (MEA, 2003), which provides a practical, tractable, and sufficiently flexible classification for categorizing the various types of ecosystem goods and services (EGS). In this context, all EGS can be generally classified into four main categories, i.e. *provisioning, regulating, cultural and supporting services* – see Table 2.6.

Table 2.6: *A general classification of Ecosystem Goods and Services for European Forests*

Types of Ecosystem Services	Examples

Supporting Services	Provisioning Services	Source of production inputs for a number of forestry sectors: wood pulp, industrial roundwood, recovered paper, sawnwood, wood-based panels, paperboard, woodfuel and other substantial goods, such as food and fruits, for supporting local livelihoods.
	Regulating Services	Climate regulation, water regulation, erosion regulation, etc.
	Cultural Services	Recreation and ecotourism, aesthetic values, spiritual and religious values, cultural heritage values, etc.

Source: adapted from MEA 2003

Provisioning Services

In this forest service category, we divide the forest products into seven main groups, including industrial roundwood, wood pulp, recovered paper sawnwood, wood-based panels, paper and paperboard, and wood fuel. For all products quantity information on the total annual removal from forests is available on the FAOSTAT-Forestry. We first collected quantity information for all 32 European countries under consideration - see Table 2.7. The physical quantification of provisioning services in turn, will be at the basis of the economic valuation exercise.[1]

***Table 2.7**: Total wood forest products provided by forestry sectors in Europe*

Country	Wood pulp (Mt/yr)	Industrial Roundwood (Million M^3/yr)	Recovered paper (Mt/yr)	Sawn-wood (Million M^3/yr)	Wood-based Panels (Million M^3/yr)	Paper-board (Mt/yr)	Woodfuel (Million M^3/yr)
Albania	0	0.08	0	0.1	0.04	0	0.22
Austria	1.93	12.79	1.42	11.07	3.45	4.95	3.69
Belgium	0.51	4.3	2.14	1.29	2.8	1.9	0.65
Bosnia&Herzegovina	0.02	2.44	0	1.32	0	0.08	1.36
Bulgaria	0.14	3.18	0.08	0.57	0.35	0.33	2.68
Croatia	0.1	3.11	0	0.62	0.13	0.59	0.91
Czech Republic	0.75	14.29	0.48	4	1.49	0.97	1.23
Denmark	0	1.03	0.44	0.2	0.35	0.42	1.26
Estonia	0.07	5.5	0.05	2.2	0.41	0.07	1.3
Finland	11.13	47.12	0.6	12.27	1.99	12.39	4.48
France	2.5	31.62	5.95	9.95	6.4	10.33	2.8
Germany	2.88	50.91	14.41	22.12	16.98	21.68	6.04
Greece	0	0.52	0.35	0.19	0.87	0.53	1
Hungary	0	2.8	0.37	0.22	0.67	0.57	3.14
Ireland	0	2.63	0.44	0.89	0.88	0.05	0.02
Italy	0.52	2.69	5.49	1.59	5.61	10	5.36
Latvia	0	11.89	0.06	4.23	0.43	0.04	0.95
Lithuania	0	4.92	0.08	1.5	0.4	0.11	1.13
Luxembourg	0	0.26	0.06	0.13	0.45	0	0.01
Netherlands	0.12	0.82	2.46	0.28	0.01	3.47	0.29
Norway	2.46	8.49	0.44	2.33	0.58	2.22	1.18
Poland	1.05	28.53	1.2	3.93	6.74	2.73	3.41

1 The data report from FAOSTAT does not provide an efficient collection of data on non-wood forest products, for this reason, our figures of the forest provisioning services will not embed this provisioning service. We acknowledge that our estimation is underestimated compare to other studies in the literature, if there is less evidence to link the provision of with non-wood forest products climate change (e.g. Merlo and Croitoru, 2005).

Portugal	1.93	10.51	0.6	1.01	1.31	1.58	0.6
Romania	0.16	11.54	0.3	4.32	1.01	0.37	2.96
Serbia&Montenegro	0.02	1.32	0	0.5	0.07	0.23	1.85
Slovakia	0.61	9.01	0.21	2.62	0.61	0.86	0.3
Slovenia	0.15	1.79	0	0.46	0.41	0.56	0.94
Spain	1.97	13.35	4.32	3.66	4.84	5.7	2.18
Sweden	12.11	91.7	1.57	18	0.75	11.74	7
Switzerland	0.26	3.98	1.24	1.59	0.97	1.75	1.07
Turkey	0.23	11.2	1.02	6.45	4.77	1.15	4.98
United Kingdom	0.34	8.27	7.76	2.86	3.4	6.24	0.32

Source: FAOSTAT, 2005

Regulating Services

As far as regulating service is concerned, two types of ecosystem services are of particular importance provided by European forests: (1) climate regulation (i.e. carbon sequestration) and (2) water and erosion regulation (i.e. watershed protection). It is important to note that we will focus only on the carbon service due to the lack of understanding of the complex relationships involved between forest biodiversity, forest area, water and erosion regulation. For this reason we shall proceed with evaluating biodiversity benefits provided by forest systems by evaluating the capacity of forests ecosystem in mitigating climate change by storing carbon in forests and its soil. Table 2.8 reports the carbon stocked by European forests (FRA, 2005).

Table 2.8: *Total forest area, forest area designated for recreational and passive uses and stocked carbon in forest ecosystems in Europe*

Country	Forest Area (1000 ha)	Carbon Stock (Mt)	Forest Area designated for recreation (1000 ha)	Forest Area designated for passive use (1000 ha)
Albania	794	62.6	62	80
Austria	3,862	937.5	301	393
Belgium	667	72.8	52	68
Bosnia and Herzegovina	2,185	177.9	210	274
Bulgaria	3,625	274.8	283	369
Croatia	2,135	575.0	166	217
Czech Republic	2,648	712.2	206	270
Denmark	500	60.9	39	51
Estonia	2,284	304.9	178	232
Finland	22,500	1,040.1	1,757	2,295
France	15,554	1,702.2	1,214	1,586
Germany	11,076	1,257.5	865	1,129
Greece	3 ,752	293.2	293	382
Hungary	1,976	515.0	154	201
Ireland	669	71.3	52	68
Italy	9,979	1,315.5	779	1,017
Latvia	2,941	392.2	229	299
Lithuania	2,099	274.6	163	214

Luxembourg	87	23.5	6	8
Netherlands	365	52.1	28	37
Norway	9,387	1,770.7	733	957
Poland	9,192	2,446.8	717	937
Portugal	3,783	161.0	295	385
Romania	6,370	1,719.5	497	649
Serbia and Montenegro	2,694	215.7	210	274
Slovakia	1,929	518.8	150	196
Slovenia	1,264	334.6	98	128
Spain	17,915	987.4	1,399	1,827
Sweden	27,528	3,597.2	22,198	2,807
Switzerland	1,221	294.6	95.36	124
Turkey	10,175	818.5	794.67	1,037
UK	2,845	409.3	222.19	290

Source: adapted from FRA, 2005

Cultural Services

In Europe, forests are of particular importance in many countries in terms of cultural services. Among all others, recreational service represents the most important value (MCPFE 2007), including hunting, natural park visiting, forest landscape and other spiritual uses. Some of the services always involve both consumptive (e.g. consumption of animal meat) and non-consumptive (e.g. enjoyment derived from hunting activities and forest landscape) uses of forests. To avoid double counting, we refer cultural services to non-consumptive use of forests only. In addition, the passive use value of the forests has an essential role in assessing some particular forest areas. In the present study, we use forests areas that are designed to recreational and protective purposes, as described by the Global Forest Resources Assessment 2005 (FRA, 2005), as key variables when assessing the welfare changes in terms of changes in the provision of cultural services. Table 8 shows the forest areas that are preliminary designated for recreational and passive uses in all selected European countries, as well as the total carbon stocked in these countries' forests.

Supporting Services

Finally, with respect to the supporting service, indicators for measuring the respective forest ecosystem changes in response to climate change are not well developed and thus quantity data to measure them are not readily available (MEA 2005). For this reason, we will not directly tackle the valuation study for this service category. However, it is important to realize that the relevant values are implicitly reflected in the valuation of all other three categories of forest ecosystem goods and services.

2. Economic valuation of the ecosystem goods and services provided by European forests

The hybrid valuation model is used to capture the values of three types of ecosystem services under consideration. First of all, for the provisioning services provided by European forests, we can infer that the economic values are the direct use values obtained from trading wood forest products in the market. Therefore, market prices are used to value this ecosystem service and its information is derived from Food and Agriculture Organization of the United Nations database (FAOSTAT) on forests. Second, in order to estimate the welfare changes associated to the carbon regulation we shall be using the avoided damage cost methods that were undertaken by the recent EC funded project, CASES[2] to estimate the marginal damage cost of per additional unit of CO_2 emission. Economic theory tells us the optimal emission level is determined by the intersection of the marginal damage cost of emissions and the marginal benefit from damage mitigation (or marginal abatement costs). Thus the crossing point is corresponding to the unit value of carbon sequestration, which gives rise to the optimal policy to incentivize the necessary abatement for achieving the global carbon stabilization goal, and can be used to calculate the total economic value of carbon stored in forests. Finally, with respect to the cultural service, meta-analysis and value transfer methods are jointly used. These two methods are anchored in non-market valuation methodologies and rely on the existing databases[3] of non-market valuation studies for forests in Europe. All values are expressed in 2005 US$. However, the specific nature and availability of data as well as the different valuation procedures embraced according to the nature of the ecosystems services under consideration will merit a separate discussion.

The Economic Value of Provisioning Services

Both total values of provisioning services of forest are estimated using the data derived from FAOSTAT. The total export values of WFPs are used as a proxy of total economic benefits derived from the annual removal of forest products. The export values used here are published by FAOSTAT in year 2005. The export values of all 7 forestry sectors are collected and summed up across all the 7 selected forestry sectors at country level and then divided by the forest size of the country – see equation (1). This gives rise to the productivity values (in $/ha term) of the forest biomes in terms of the profits associated with the types of WFPs to be delivered to the market (see Table 5). Note that the productive values vary across countries as they reflect the different contributions of various forest biomes to the national economies.

$$ProductivityValue_i^{WFPs} = \sum_{n=1}^{7} ExportValue_{ni} \Big/ ForestArea_{ni} \qquad (1)$$

where *i* refers to one of the European countries and *n* is the type of WFPs under consideration.

The Economic Value of Regulating Services

2 CASES stands for "Cost Assessment of Sustainable Energy Systems" for EU countries and the selected non-EU countries, including Turkey, Brazil, India and China. The study aimed at providing a comprehensive and dynamic assessment of the full costs of electricity generation based on the state-of-the art methodologies, taking into account both geographical and temporal extend of the impacts and social economic impacts, such as health and safety, economic production and consumption, recreation, and environmental and natural assets caused by climate change.

3 The popular databases for non-market valuation study include: Environmental Valuation Reference Inventory (EVRI), Envalue, and the Ecosystem Services Database.

Forest conservation or prevention of deforestation in order to stabilize Green House Gas (GHG) emissions – questions not originally included in the Kyoto Protocol – have been officially recognized in COP13 in Bali in December 2007 as important issues. The estimation of economic value of climate regulating services (i.e. carbon storage) provided by forest ecosystem is therefore considered to have very important impacts on policy making for CO_2 stabilization in Europe. However, it is important to note that our economic value estimates for regulating service in the present paper are underestimated, as the present economic valuation exercise focuses only on carbon sequestration services provided by forests. Nevertheless, further investigation will lead to a much higher value estimates of this service, by taking into account other regulating services, e.g. watershed protection and soil nutrient cycling.

The methodological framework for valuing the regulating services consists of two steps: we first compute the marginal value of carbon storage in forests (2005US$/tC), which will then be used to estimate the total economic values that can be obtained in different forest ecosystems across Europe. The marginal value of carbon storage refers to the benefits from avoided damages[4] caused by incremental of CO_2 or CO_2-equivalent GHG emissions in the atmosphere due to the carbon sequestration function of forest ecosystem. In the present paper, we built our analysis upon an existing project, "Cost Assessment for Sustainable Energy Systems" - CASES[5], funded by EU but targeting at a worldwide study. One of the main features of CASES is that it is built upon the Integrated Assessment Models (IAMs), which by definition combine the dynamics of global economic growth with the dynamics of geophysical climate dynamics, to estimate the cost of GHG emissions under different energy evolution paths in 2020, 2030 and 2050. The existing literature on IAM has been used intensively reviewed under the project and various available estimates in the recent years were taken into account in its finally delivered value estimates. Among all others, the value of social costs of carbon estimated by UK's Department for Environment, Food and Rural Affairs (DEFRA 2005) was adopted for it is reflexive to the policy context in which the values are used, and it combines the results of a number of IAM's in a transparent matter. As a consequence, CASES project was able to obtain three levels of estimates of marginal damage costs, i.e. lower, upper and central estimates[6], respectively. For instance, as reported in CASES final report, the lower estimates of marginal damage costs of carbon (also known as the social costs of carbon) renage from € 4/tCO_2 in 2000 to € 8/tCO_2 in 2030; the upper estimates evolve from € 53/tCO_2 in 2000 to € 110/tCO_2 in 2030; and the central estimate evolves from € 23/tCO_2 in 2000 to € 41/tCO_2 in 2030.

For the present valuation exercise, the lower bound estimate of the social cost of carbon is adopted. The original value estimates was adjusted by discounting it to the real Euro value in

---

4 The avoided damage costs assessment method has been widely used in the literature (see Cline, 1992; Nordhaus, 1993a,b; Merlo&Croitoru, 2005; CASES, 2008) to calculate indirectly the benefits from carbon sequestrated in forests, but it is important to note that the concept is different from the market price of carbon (obtained via emission trading scheme) and the marginal abatement cost (involves the costs of technological R&D for facilitating the emission abatement), although under certain restrictive assumptions the three measures would be broadly equal, at the margin (DEFRA, 2007).
5 CASES, Project No.518294 SES6, (2006-2008).
Project official website: http://www.feem-Project.net/cases/
6 The values are based on full Monte Carlo runs of the FUND and PAGE models, in which all parameters varied to reflect the uncertainty surrounding the central parameter values in both models. The lower and upper bounds are the 5% and 95% probability values of the PAGE model, while the central guidance value is based on the average of the mean values of the FUND and PAGE models. A declining discount rates is use as suggested by the UK Government 'Green Book'. The equity weighting of damages in different regions is applied to aggregate the regional damage costs to global damages, in other words, damages in richer regions receive lower weights and damages in poorer regions receive higher weights.

2005 at a discount rate of 3%, and was then converted to 2005US$ based on the real exchange rate of the year and the Purchasing Power Parity (PPP).

The value of cultural services

The cultural services provided by forest ecosystems consist of two components in our analysis: recreational use (e.g. nature-based *tourism in forests*) and passive use (e.g. existence and bequest value of forests and biodiversity). Not being traded in regular markets, recreation and passive use values are usually measured as willingness to pay (WTP) figures derived from revealed or stated preference valuation techniques, such as travel cost method, contingent valuation and choice experiment, etc. According to previous literature reviews on cultural values, a simple expected utility specification can be used to describe how individuals are willing to trade off wealth for increases or decreases in forest cultural services, under the assumption that the estimated marginal value of the service decreases with an increase in the size of the forest site, and increases with an increase of the income level of the country where the forest is located (e.g., Hammitt, 2000; Markandya *et al.,* 2008). The changes in future forest areas are driven by various forces, including the current concern of climate change. Under such circumstance, the expressed WTP estimate for trading off the forest resources also reflects the fact that individual's preference to enjoy a certain kind of culture service may shift from one forest to another driven by the changed future climate conditions.

Due to the large scale of our study, we focus on estimating the average WTP (expressed in 2005$/ha) for the cultural services (either recreational use or passive use) from various forest ecosystems across Europe. Value transfer has been conducted between the study sites, where original valuation has been carried out and the targeted policy sites, where forest biomes are found similar to those in the study sites.

Running a meta-analysis is the first step. The result of meta-analysis enables us to explain the variance of the available WTPs (Willingness-To-Pay) as a function of a few statistically significant explanatory variables[7]. In particular, main explanatory factors for forest recreation and passive use are: i) size of recreational forest sites; and for passive use, size of forest areas designated to biodiversity conservation; and ii) income level in the study area. The WTP figures included in the regression are selected from an extensive literature review process focusing on all existing valuable studies. The estimated coefficients are then used for the geographical value transfer across countries. The selected primary studies covering various types of forest biomes in Europe are presented in Table 2.9.

Table 2.9 *Selected primary valuation studies at the study sites for geographical value*

7 A similar approach is used by the authors in another recent research project (COPI) concerning a worldwide valuation of forest ecosystems in the context of policy inaction rather climate change (see Markandya et al. 2008 for more details).

transfer

Country	Reference study	Forest biome
United Kingdom	Garrod, G.D. and Willis, K. G. (1997)	Temperate forests
	Hanley, N., Willis, K, Powe, N, Anderson, M. (2002)	
	ERM Report to UK Forestry Commission (1996)	
Finland	Kniivila, M., Ovaskainen, V. and Saastamoinen, O. (2002)	Boreal forests
	Siikamaki, Juha (2007)	
Spain	Mogas, J., Riera, P. and Bennett, J. (2006)	Mediterranean
United Kingdom	Scarpa, R., S. M. Chilton, W. G. Hutchinson, J. Buongiorno (2000)	Temperate broadleaf and mixed forests
The Netherlands	Scarpa, R., S. M. Chilton, W. G. Hutchinson, J. Buongiorno (2000)	Temperate broadleaf and mixed forests
Finland	Bostedt, G. and L. Mattsson (2005)	Boreal forests
Italy	Bellu, L. G. and Cistulli V. (1994)	Mediterranean and Temperate Broadleaf

At the second step, the WTP estimates obtained in the selected primary studies[8] are transferred to the policy sites, i.e. other European countries where original valuation studies are absent. The value transfer function is shown in equation (2) below, where the coefficients of forest size (*S*) and per capita income (*PPP_GDP*) obtained in previous meta-regression analysis are used to correct the transferred WTP estimates. In addition, the number of households (*H*) is also in the value transfer function, as it is considered to be influential on the final WTP estimates in different regions. WTP estimates from recreational and passive uses of forests are transferred separately for each country and a sum of the two values give rise to the WTP for the cultural value of forests in the country.

$$V_i = V_j\left(\frac{H_i}{H_j}\right)\left(\frac{S_j}{S_i}\right)^{\beta}\left(\frac{PPPGDP_i}{PPPGDP_j}\right)^{\gamma}$$

where:
V_i = estimated *WTP*/ha/year for a country *i*
i = policy site, which refers to countries where value estimates of WTP is needed, but primary studies are absent.
j = study sites, where original valuation studies are found.

Finally, by multiplying the WTP estimates *V*($/ha) for recreational or passive use of forests by the sizes of forest area *S* that have been designated for recreation or conservation, we can obtain the total recreational or passive use value for each country. The total cultural value of a European country refers to the sum of the respective recreational and passive use value of the forests.

8 When several representative case studies and values are available, the mean marginal value is used.

Final valuation results

In summary, the estimated economic values of three ecosystem services, i.e. regulating, cultural and provisioning services are presented in the Table 2.10, all expressed in 2005 US dollars.

Table 2.10 Economic values derived from three forest ecosystem services in Europe

Country	Regulating Service (2005 Million US$/yr)	Cultural Services (2005 Million US$/yr)	Provisioning services (2005 Million US$/yr)	Total (2005 Million US$/yr)
Albania	305	0.3	6	1,300
Austria	4,451	91	5,990	24,949
Belgium	344	75	4,807	6,339
Bosnia&Herzegovina	839	0.2	202	3,761
Bulgaria	1,393	40	256	6,200
Croatia	2,721	8.2	343	11,884
Czech Republic	3,375	73	1,568	15,946
Denmark	296	57	465	1,776
Estonia	1,465	2.3	510	6,723
Finland	4,913	3.3	12,067	32,897
France	8,137	831	7,204	42,529
Germany	5,933	2,440	16,636	44,228
Greece	1,442	89	141	6,341
Hungary	2,518	107	693	11,474
Ireland	370	0.03	506	2,072
Italy	6,557	1,734	3,225	32,753
Latvia	1,887	1.1	977	8,976
Lithuania	1,347	7.8	354	6,069
Luxembourg	111	5.2	216	691
Netherlands	249	166	3,693	4,915
Norway	3,744	1.2	1,863	17,737
Poland	11,714	224	2,127	52,007
Portugal	802	42	1,859	5,302
Romania	8,118	143	848	35,403
Serbia&Montenegro	1,035	0.3	137	4,525
Slovakia	2,458	35	1,025	11,481
Slovenia	1,611	17	684	7,529
Spain	5,078	1,034	3,337	25,897
Sweden	8,371	149	13,200	48,834
Switzerland	1,416	46	2,003	8,050
Turkey	3,909	0.02	256	16,827
United Kingdom	1,967	734	2,665	11,739

Table 2.10 shows the weight of ecosystem service value in a country's total received forestry benefits may vary depending on the type and extent of the forests in the country as well as the ecosystem services under consideration. Finally, the last column of the table calculates the aggregated economic value that each European country can get from their forest ecosystems.

Not surprisingly, highest aggregated economic values are mostly found in forests located in Central - Northern European countries where host (a) the largest forest areas, (2) the higher number of households, (3) high rates of forest recreationists among the households. In addition, high values are found also in two eastern European countries, i.e. Poland and Romania, due to the rich forest resources and large forest areas found in these countries. For an aggregated perspective, we can see that half of the biodiversity benefits from the European forests are mainly concentrated in the regulating services, which count for about half of the total value. Cultural values amount to 5% and the provisioning services 45%. In addition, if we take a closer look into the cultural value component, we can see that Germany, Italy, Spain, France, UK and Poland are the countries that show the highest economic values on this component. However the relative value composition is not the same among those countries – see Figure 2.5.

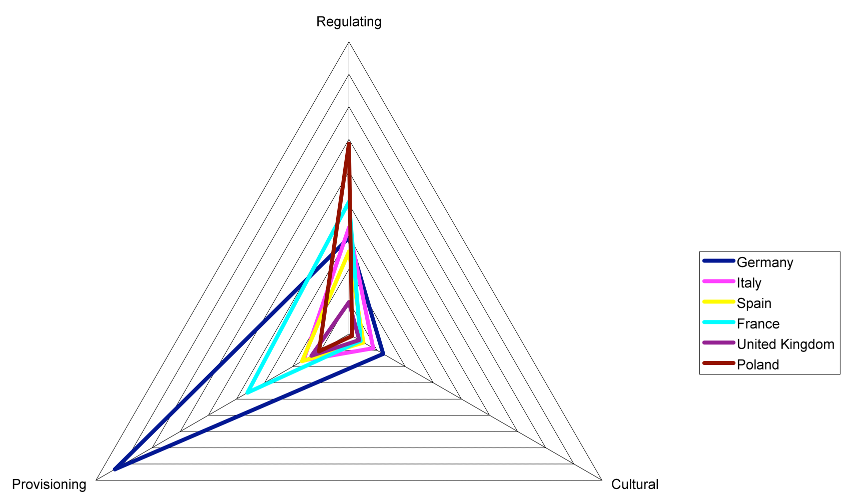

Figure 2.5: Forest ecosystem services composition

As we can see, in Germany, forests are predominantly producing provisioning services. In fact, in this setting Germany is the country that has the strongest profile of provisioning services. On the other hand, Poland has the weakest profile in forest provisioning services. However, Poland has the strongest profile with respect to regulating services. Italy, France and Spain do not have any predominant profile with respect to any of the forest ecosystem services but show the strongest balance in terms of distribution of the economic value for each of the three dimension under consideration. Finally, the UK show a profile closer to the France/Spain/Italy rather than Germany or Poland. In any case, the intensity of the forest values produced in the UK when compared to France/Spain/Italy are weaker: the UK profile lies inside all the individual maps of France/Spain/Italy.

In addition, we can also explore the use of GIS tools so as to map the economic values of forest ecosystem services – see Figure 2.6.

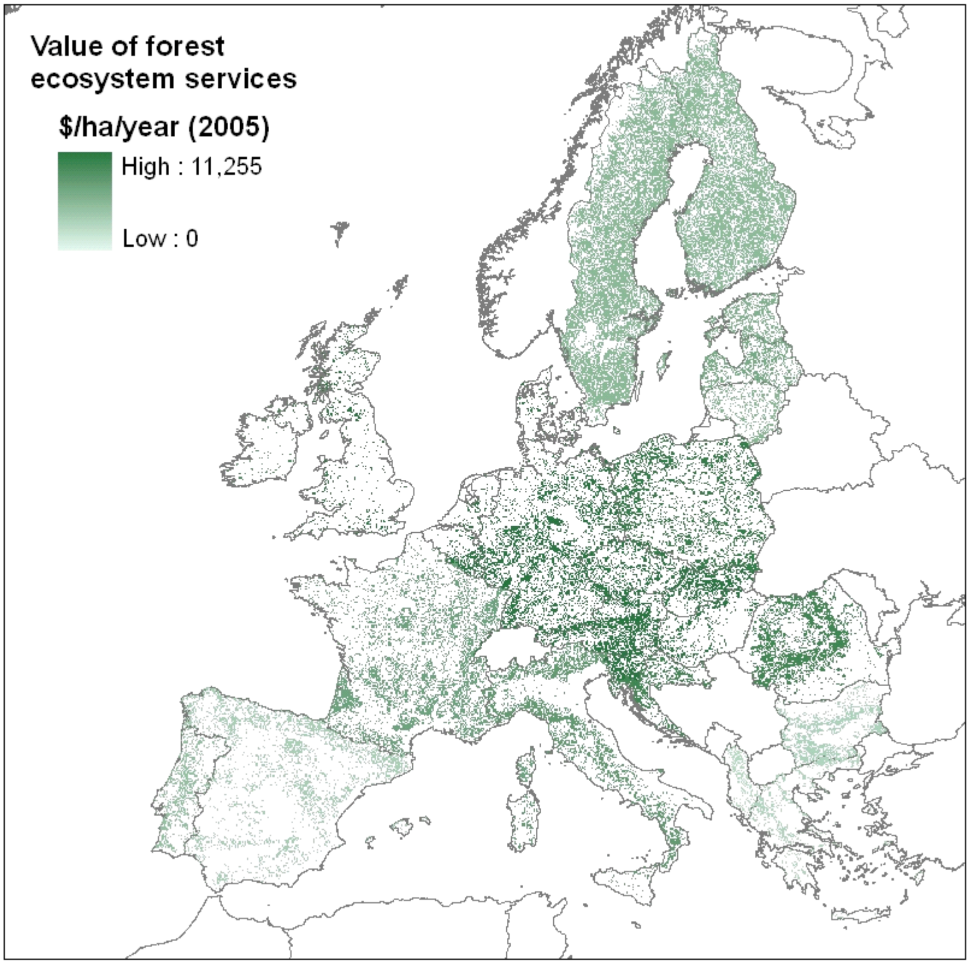

Figure 6: The productivity value of per hectare forests in Europe

The GIS map is created based on the geographical distribution of forests in Europe as identified in the Corine land use map. Within each country, the average per hectare values estimated in the economic valuation analysis described in this section are distributed over the forest grid cells in Corine, with a 100x100 meter resolution. These maps provide detailed information with respect to the spatial distribution of the economic values. This can be also integrated with the chart diagrams as the one showed in Figure 2.7. Whereas Spain and France show again similar profiles, which are characterized by a balanced distribution of the values at stake, respectively provisioning, regulating and cultural, UK reveals to be the country with the highest forest productivity in terms of cultural values. Germany and Italy reveal to be the second and third most productive European forests, again when measured in terms of cultural values.[9] As far as the regulating services, all the countries show similar profiles where the differences account the differences of the forest type and geographical locations.

9 The Netherlands is the most productive country in terms of cultural values provided by forests, well ahead the UK.

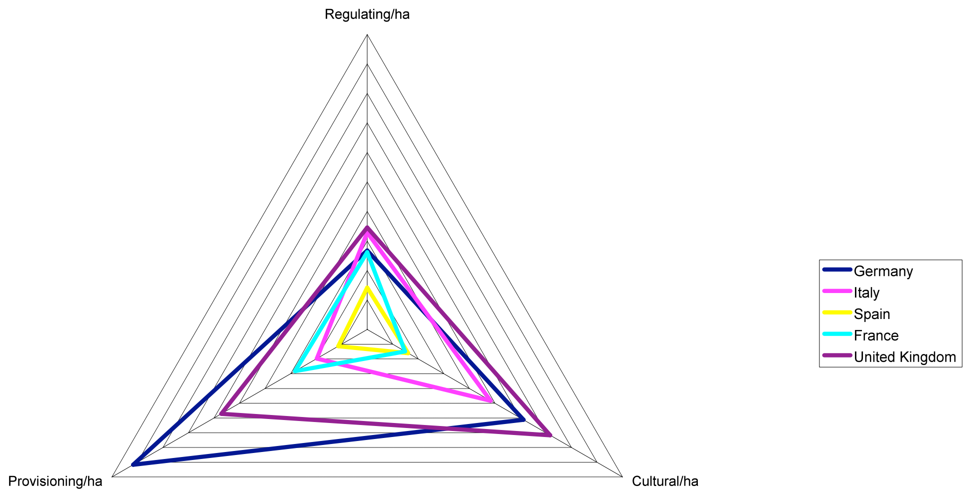

Figure 2.7: The productivity value of per hectare forests in Germany, Italy, Spain

3.2.2 Economic Value of Freshwater Ecosystems

Freshwater ecosystems have long been recognized as sources of important services and goods for humans. The range of benefits encompasses provisioning of goods such as water, fuel wood, materials, and fish for commercial exploitation, regulating flood events and water quality processes, providing the setting for recreational activities and amenity values, and supporting a rich biological diversity. Both the level of provisioning of ecosystem services and their impact on human welfare are threatened by a series of environmental stressors, such as habitat conversion and climate change, which have a potential to affect the ecological equilibriums services rely upon and the patterns of human exploitation.

In this study we develop a framework for the economic valuation of the flow of services from freshwater ecosystems (i.e., rivers, lakes and freshwater wetlands) at European scale and for assessing the distribution of such values within European countries. Meta-analysis is used as a tool (i) to investigate the provision of services of freshwater ecosystems from an economic perspective and (ii) to scale up freshwater ecosystem services values in 28 European countries. In particular, we investigate how climatic conditions, socio-economic characteristics of the population living in the surrounding of the valued sites, and biodiversity richness are mutually linked in determining the impact on human livelihood produced by the provision of ecosystem services. The aggregated values are subsequently geographically distributed within European countries based on the information on land cover provided by the Corine map (Büttner et al. 2002).

1. The dataset of freshwater ecosystems valuations

The impact of freshwater ecosystem services on human livelihood is predicted based on 236 independent observations from 103 valuation studies and concerning 123 distinct freshwater wetlands, riverine or lacustrine ecosystems. The dataset is not limited to European sites but includes valuation results from ecosystems worldwide. Figure 2.8 illustrates the geographical distribution of the ecosystems in the dataset.

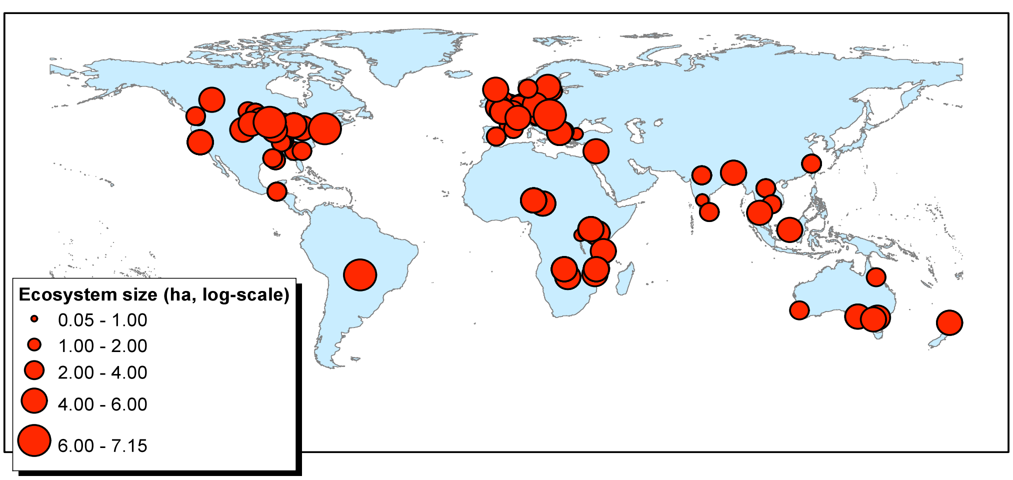

Figure 2.8 Geographical location of valued ecosystems and their size

All continents are represented in the data set. The largest number of observations pertains to North America (90) and Europe (63), but a significant fraction comes from Africa (37) and Asia (32). Australasia (8) and South America (6) are somewhat underrepresented.

All studies considered are primary valuation studies and no observation based on value transfer is included in the data set. In order to limit the risk of introducing publication bias, the investigation is not limited to the analysis of publications in the "official scientific literature", but also explores the "grey literature" (such as reports for both public and private institutions and consultancy studies) and unpublished research results. Efforts were also made to include studies that are not published in the English language. The average number of observations per study (2.3) and the maximum number of observations for a single study (12) are relatively low if compared to the total number of observations used in the analysis (236).

2. Specification of the meta-regression model

The meta-analytical regression model used in this study is a semi-logarithmic model where the dependent variable is the values standardized to 2003 US$ per hectare per year and corrected for purchase power parity across different countries. The explanatory variables are selected to represent characteristics of the valuation study, the specific valued ecosystem site and the socio-economic, geographical and climatic context in which the valued ecosystem is located.

Study variables

Study characteristics accounted for in the model include the valuation method used and a dummy distinguishing between marginal and average values. A range of valuation methods has been used in the primary studies for the assessment of the values of wetlands and freshwater ecosystems. Valuation methods are grouped in four categories: stated preference methods (i.e., contingent valuation method and choice experiment), revealed preference methods (i.e., travel cost method and hedonic pricing), market-based methods (i.e., market prices, replacement cost, net factor income, production function and opportunity cost), and mixed valuation methods, which combine different methodologies (e.g. contingent behaviour method). A dummy for each of the categories is included in the meta-regression model to account for the heterogeneity of methods, as they produce estimates of different welfare measures and not all of them have a strong basis in welfare theory. Market-based values are the omitted variable in the regression. Figure 2.9 illustrates the distribution of valuation methods according to the four categories of methods.

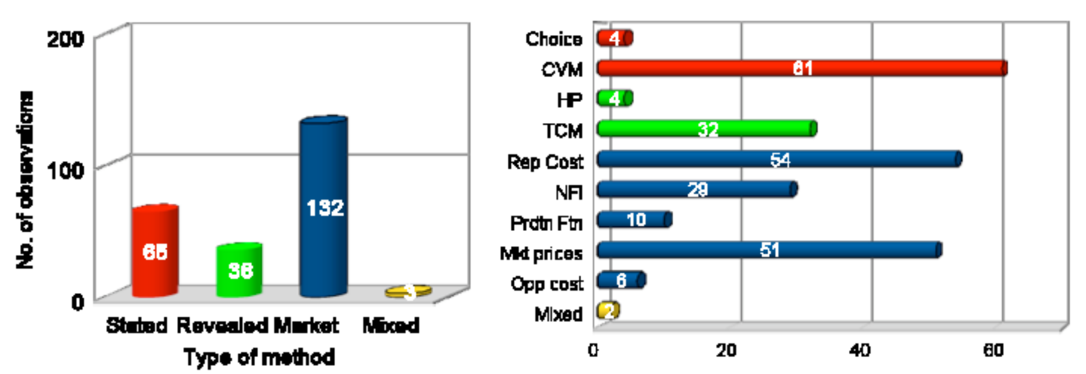

Figure 2.9 Number of observation per type of method and single valuation methods (Choice = choice experiment, CVM = contingent valuation method, HP = hedonic pricing, TCM = travel cost method, Rep Cost = replacement cost, NFI = net factor income, Prdtn Ftn = production function, Mkt price = market prices, Opp cost = opportunity cost)

The largest number of observations was derived using market-based valuation methods (132). The most frequently implemented market-based method is market prices (51) followed by net factor income (29). Production function and opportunity cost were used in respectively 10 and 6 observations. Some studies use a combination of market-based methods. Stated preference methods were used in 65 observations, contingent valuation method (61) being more frequently implemented than choice experiment (4). Revealed preference methods (i.e., travel cost method (32) and hedonic pricing (4)) were used in 36 observations.

To distinguish between marginal and average per hectare values, a dummy variable is introduced, which assumes a value equal to one for marginal values (34) and equal to zero otherwise (202).

Site variables

Characteristics of the valued ecosystem that are accounted for in the meta-regression model are the type and size of the ecosystem, and the services provided. Three dummies identifying rivers and floodplains (106 observations), lakes (89 observations) and other types of freshwater wetlands (93 observations) are introduced in the model. Some value observations may pertain to several ecosystem types.

Three types of freshwater ecosystems were considered: rivers (and river floodplains), lakes, and other types of freshwater wetlands, such as palustrine wetlands, swamps, peat bogs and wet forests. River deltas, estuaries, coastal salt marshes and lagoons were not included in the dataset. Figure 2.10 illustrates the distribution of the observations across the three main ecosystem types considered. Since a value estimate may pertain to an ecosystem with mixed characteristics of to a group of ecosystems of different types, the sum of observations for the categories in Figure 2.10 is larger than the total number of observations.

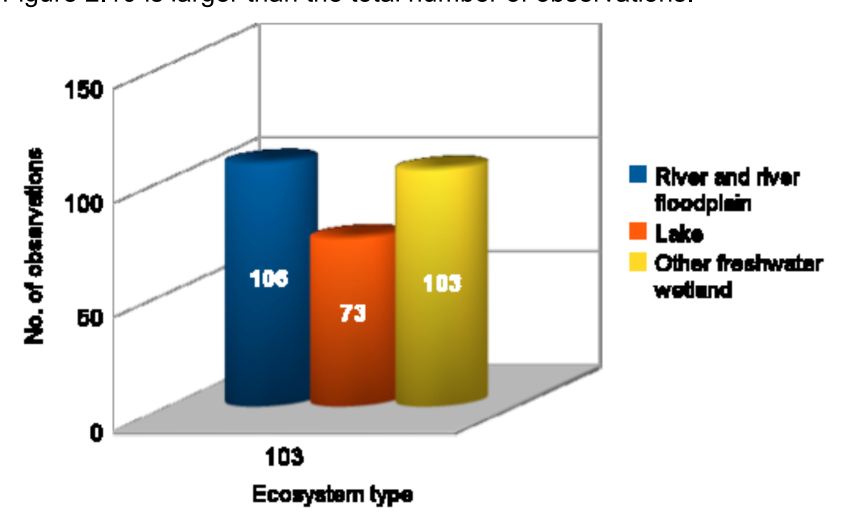

Figure 2.10. Number of valuations for rivers, lakes and freshwater wetlands ecosystems

The services and goods provided by the investigated ecosystems are classified according to the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) approach into the categories of provisioning, regulating and cultural services. Within the category of cultural services a distinction is made between recreational services (i.e., recreational hunting, recreational fishing, and other non-consumptive recreational activities such as walking, cycling, swimming and boating) and passive uses (i.e., amenity value and provision of natural habitat and biodiversity). The number of observations for the identified ecosystem services is illustrated in Fig. 6. The largest number of observations is for recreational cultural services as well (98). A relatively large number of observations are available for provisioning services (88) such as commercial fishing and hunting, harvesting of natural materials, water supply and fuel wood. Regulating services such as flood control and storm buffering and water quality improvement provided by 63 observations. Slightly less information is available in the literature for passive uses (54 observations).

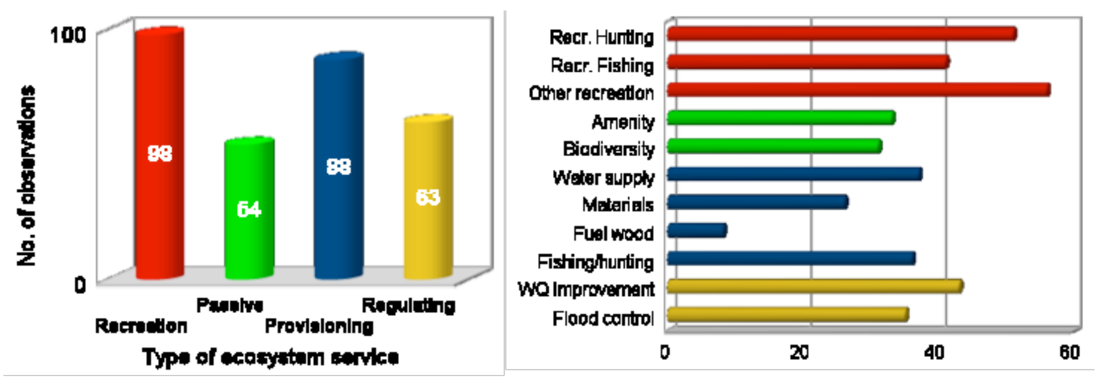

Figure 2.11. Number of observation per type of ecosystem service and single services (Recr. Hunting = recreational hunting, Recr. Fishing = recreational fishing, Fishing/hunting = commercial fishing and hunting, WQ improvement = water quality improvement, Flood control = storm buffering and flood control)

The size of the ecosystems has been estimated in hectares and shows large variability. The ranked distribution in the size of the valued ecosystems is illustrated in Figure 2.11. The median size is 3,455 ha, while the average size is 187,875 ha with a standard deviation equal to 1,299,594 ha. Examples of large valued sites, covering hundreds of thousands of hectares are the wetlands of Louisiana (Gosselink et al. 1974), the Pantanal (Shrestha et al. 2002), and the Danube floodplain (I. M. Gren et al. 1995). Although the majority of the valuation studies so far has comprehensively focused on large sites, small-size ecosystems are also represented. Some examples are small wetlands in the North Dakota prairie (Leitch & Hovde 1996), Louisiana (Leitch & Hovde 1996), Italy (Marangon et al. 2002) and England (Ledoux 2003). All these wetlands are below hundred hectares in size. Although there is no clear *a priori* expectation of the influence of size on its value, previous meta-analyses of ecosystem values agree on the relevance of size as a significant factor to explain the variability of values.

Finally, the latitude at which the valued ecosystems are located is included in the model as a categorical predictor with four levels. The four categories considered are chosen so as to distinguish between different biomes in European countries: Mediterranean (between 35°N and 45°N), temperate (between 45°N and 55°N), and Baltic-Scandinavian ecosystem types (at latitudes higher than 55°N). The omitted category in the regression identifies freshwater ecosystems that are located at latitudes equal to or lower than 35°N.

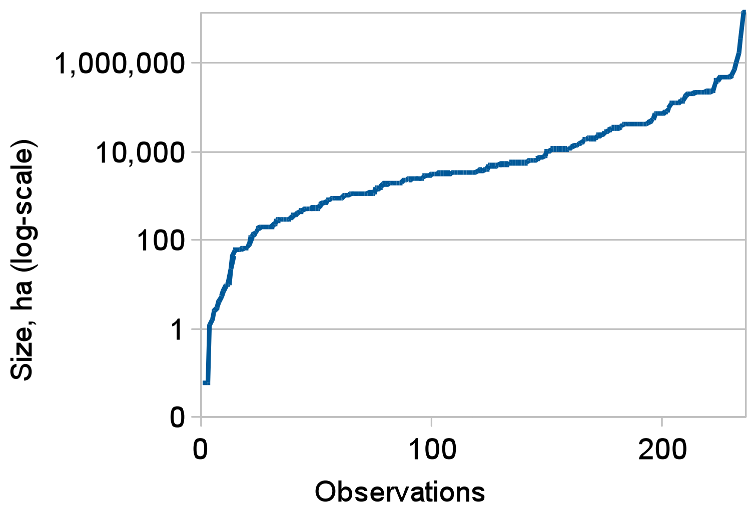

Figure 2.12 Ranked distribution in the size of the valued ecosystems

Context variables

Environmental valuation studies carried out at different geographical sites and involving populations with different socio-economic characteristics and consumer preferences generally produce different outcomes (Brouwer 2000). Context characteristics are expected to significantly influence the valuation estimates.

A series of context variables are included in the meta-regression model. Gross Domestic Product (GDP) per capita and the population living in the surroundings of the valued ecosystem are introduced to capture some characteristics of the socio-economic context where the valued sites are located. The presence of income effects and the influence of population density in the surrounding of the valued environmental asset in determining the results of the valuation study were identified in previous meta-analyses (Brander et al. 2006). The GDP value is calculated at country level for all countries but European countries and the USA, where it is estimated at NUTS2[10] and state level respectively. GDP per capita values were expressed in Purchase Power Parity (PPP) units and standardized to I$ 2003 per year following the procedure described in (Ghermandi et al. 2008). The population density and total area of freshwater ecosystems abundance of wetland ecosystems at country level are included in the meta-regression model in order to capture the fact that a high population density around the valued sites may contribute to transform potential values into actual benefits as well as potential substitution effects (Ghermandi et al. 2008). The total area of wetlands for each country is calculated based on the georeferenced information provided in the Global Lakes and Wetlands Database (Lehner & Döll 2004).

[10] NUTS2 is one of the Nomenclature of Territorial Units for Statistics levels used in the European Union to identify regional administrative divisions within member states.

A series of geo-climatic and biodiversity variables were considered for their possible influence on the estimated values. The biodiversity variables considered include the total number of bird (mammal) species and of threatened bird (mammal) species at country level as derived from the Little Green Data Book (World Bank, The 2007). The geo-climatic variables evaluated for inclusion in the model include the average, minimum and maximum monthly temperature and the average yearly precipitation at country level. The geo-climatic and biodiversity data used in the model refers to the baseline year 2003.

Standardization of values

To allow for a comparison between ecosystem values that have been calculated in different years and expressed in different currencies and metrics – e.g. willingness to pay (WTP) per household per year, capitalized values, and marginal value per acre – standardization to common metric and currency is needed. WTP per household per year cannot be used as a common metric since several of the valuation methods used in the literature – e.g. net factor income, opportunity cost, replacement cost, and market prices – do not produce WTP per person estimates. On the other hand WTP per person can be converted to a value per hectare per year if the relevant population is known. Values were thus standardized to 2003 I$ per hectare per year. Values referring to different years were deflated using appropriate factors from the World Bank Millennium Development Indicators (World Bank 2006), while differences in purchase power among the countries were accounted for by the PPP index provided by the Penn World Table (Heston et al. 2006).

3. Econometric results

Most of the explanatory variables included in the model are statistically significant in explaining the current values of freshwater ecosystems. Of the study characteristics, revealed preference methods produce significantly lower results than market-based methods and marginal values are higher than average ones. If compared with the results of previous meta-analyses of wetland values, such result is consistent with the findings of Brander et al. (2006) and Ghermandi et al. (2008) who found high values for studies with stated preference methods and marginal values, and only partially contrasts with those of Woodward & Wui (2001), who observed high values for studies using hedonic pricing and replacement cost as valuation method.

Site-specific characteristics significantly affect ecosystem values. The coefficient on wetland size indicates decreasing returns to scale (cf. Brander et al. 2006; Ghermandi et al. 2008; Woodward & Wui 2001). Ecosystems located at temperate Northern latitudes between 35°N and 45°N provide statistically higher values than ecosystems at higher latitudes, in proximity of the Equator or at temperate climates in the Southern hemisphere. Of the ecosystem services, the coefficient of provisioning services is negative indicating low values for commercial fishing and hunting, and provision of materials and fuel wood, while both regulating services and passive values have positive – though not statistically significant – coefficients.

All context variables are significant in explaining the values of freshwater ecosystems. Both real GPD per capita and population density are positively related to ecosystem values indicating an inelastic income effect and high values where a large population may easily access the sites. The negative value on the total area of freshwater ecosystems suggests that substitution effects may take place and thus high values for ecosystems that are more unique in their environment. Such results confirm the previous findings in Ghermandi et al. (2008).

An important additional contribution of this meta-analysis is the recognition of the role of geo-climatic and biodiversity variables in determining ecosystem values. Both the coefficients of the maximum monthly temperature and total number of bird species are significant. Values tend to be high in areas of high biodiversity and decrease at high temperatures.

4. Scaling up and distribution of values

As second step in the analysis, we apply benefit transfer techniques to use the results of the meta-regression and scale up the values of freshwater ecosystem services at country level in 28 European countries. For this purpose, we evaluated the values of the explanatory variables in the meta-regression model in each of the 28 countries. Each country was classified within one of the latitude categories following the distribution of biome types within Europe. Mediterranean and South European countries include Bulgaria, Croatia, Greece, Italy, Portugal and Spain, (35°N - 45°N); Central-Northern European countries include Austria, Belgium, Czech Republic, France, Germany, Hungary, Ireland, Luxembourg, Netherlands, Poland, Romania, Slovakia, Slovenia, and Switzerland (35°N - 45°N); Northern European and Scandinavian countries are Denmark, Estonia, Finland, Latvia, Lithuania, Norway, Sweden, and the United Kingdom (latitude higher than 55°N). The context variable accounting for the total area of freshwater ecosystems in each country was evaluated by means of GIS analysis from the Global Lakes and Wetlands Database (Lehner & Döll 2004). The average size of freshwater ecosystems in European countries was derived from Brander et al. (2008), who created a dataset of 50,533 individual European coastal and freshwater wetlands with GIS analysis from the Corine land cover. The binary variables identifying valuation methods and ecosystem services were estimated at the sample mean for the scaling up of values with exception of the variable ‘marginal’ which was set equal to zero in order to estimate average values.

To aggregate the values and scale them up at country level we multiplied the per hectare values estimated for each country with the benefit transfer exercise by the total area of freshwater ecosystems in the investigated countries. Since the Corine dataset provides a more refined land use classification for European countries, the total area for the aggregation of the values was estimated based on the categories of inland marshes, peatbogs, water courses and inland water bodies in the Corine classification.

Table 2.11 presents the mean value per hectare, the total area per country and the estimated aggregated value of ecosystem services provided by freshwater ecosystems for each of the 28 European countries.

Table 2.11. Estimated value of freshwater ecosystem services in Europe

Country	Mean value [$/ha year]	Total area [ha]	Aggregated value [Million US$2003/year]
Austria	17,969	95,685	1,719
Belgium	113,286	24,762	2,805
Bulgaria	69,497	111,972	7,782
Croatia	166,508	71,551	11,914

Czech Republic	14,589	60,688	885
Denmark	11,266	90,495	1,020
Estonia	1,205	396,919	478
Finland	1,779	5,396,898	9,599
France	10,851	400,351	4,344
Germany	14,935	518,158	7,739
Greece	81,645	132,851	10,847
Hungary	5,867	279,976	1,642
Ireland	9,155	1,271,368	11,640
Italy	200,278	233,984	46,862
Latvia	2,396	272,944	654
Lithuania	1,789	182,333	326
Luxemburg	121,994	733	89
Netherlands	20,734	226,065	4,687
Norway	3,672	1,005,407	3,692
Poland	6,150	556,487	3,423
Portugal	275,265	55,567	15,296
Romania	4,495	683,155	3,071
Slovakia	12,728	30,435	387
Slovenia	30,095	10,307	310
Spain	117,314	342,307	40,157
Sweden	5,926	6,523,231	38,658
Switzerland	19,624	52,326	1,027
UK	8,819	747,987	6,596

On average, Mediterranean countries (Italy and Portugal in particular) show high mean values per hectare. This is partly due to the relative scarcity of freshwater ecosystems compared to Northern European countries. Countries with high population density such as Belgium or high values of GDP per capita such as Luxembourg also show high values. The lowest mean values per hectare are found in Scandinavian countries and Ireland, i.e., where the largest total area of freshwater ecosystems is concentrated and population density is low. We estimate thus that the highest aggregated values are in countries with high mean values per hectare, such as Italy and Spain, or with very large total ecosystem areas, such as Sweden. Despite the large area in Finland and Norway, the aggregated values for these two countries are relatively low.

To investigate the spatial distribution of values within each country, we combined the results of the meta-analysis and value transfer exercise with the information on land use from the Corine land use map. The spatial location of inland wetlands and freshwater ecosystems is identified and the average per hectare value estimated with the meta-regression is attributed to each pixel according to the country where the pixel is located. Since the resolution of the Corine map is 100 m x 100 m (i.e., every grid cell has an extension of one hectare), the value thus attributed coincides with the yearly flow of value from each grid cell. The distribution of values thus obtained is presented in Figure 2.13.

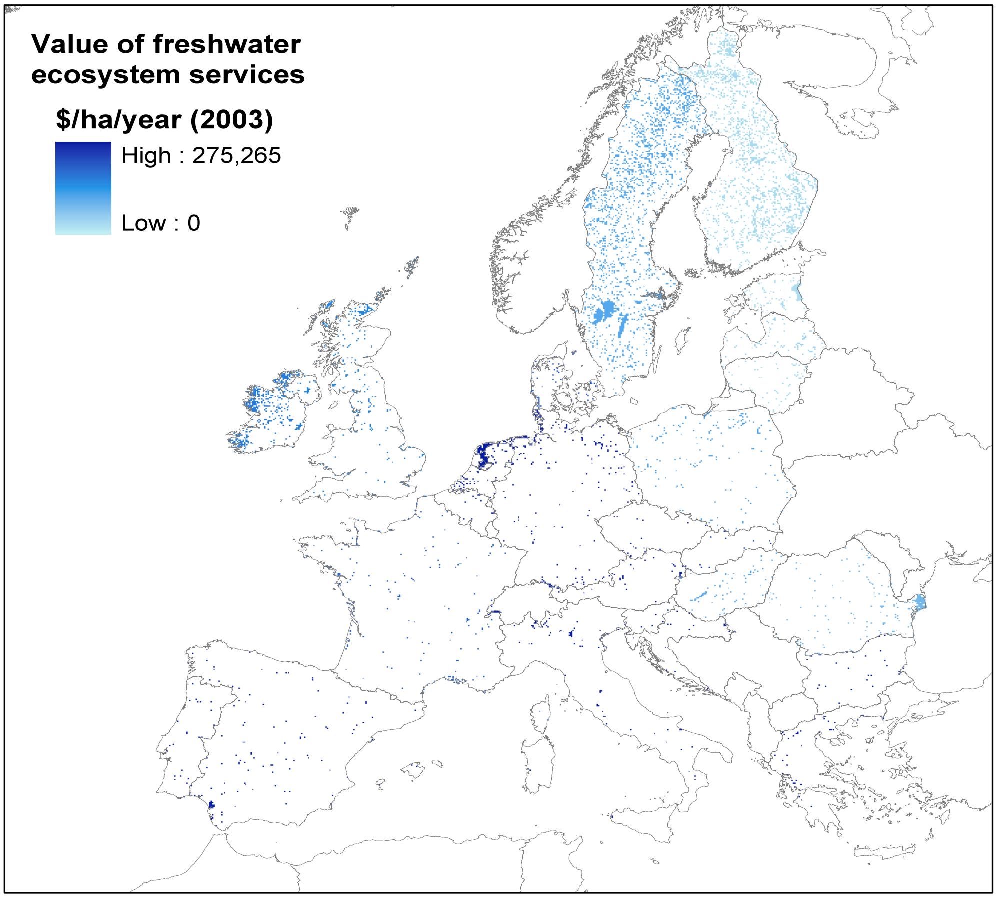

Figure 2.13. The distribution of values of freshwater ecosystem services in Europe

Figure 2.13 illustrates how the spatial distribution of economic values does not necessarily follow the geographic distribution of freshwater ecosystems. While most freshwater ecosystems are concentrated in North Europe (i.e., in Scandinavian and Baltic countries, United Kingdom, and Ireland), such sites are characterized by low per hectare values (bright blue in the map). Ecosystems with substantially higher per hectare values are scattered in Southern European and Mediterranean countries (dark blue in the map).

3.2.3 The recreational value of coastal and marine ecosystems

Marine and coastal areas host biodiversity-rich ecosystems that are among the world's most valuable. Apart from their ecological value, coastal ecosystems deliver a series of goods and services that are of benefit to humanity. These include cultural values that support tourism and recreational activities such as beach leisure (Bin et al. 2005; Freeman III 1995), wildlife watching (Loomis et al. 2000), diving (Depondt & Green 2006), bathing (Georgiou et al. 1998) recreational fishing and boating (Freeman III 1995). Market failures induced by the public good character of many of the aforementioned goods and services or from ill-defined property rights result in many of the benefits delivered by coastal and marine ecosystems being overlooked in the policy-making process. While the number of published primary valuation studies focusing on the cultural values of marine and coastal ecosystems is rapidly growing, there is still a limited understanding of what the principal drivers of coastal recreation values are and how human welfare may be affected by disappearance of habitats and species due to anthropogenic pressure and shifting environmental conditions (Brander et al. 2007; Liu & Stern 2008).

In this section we develop a meta-analytical value transfer function that will be applied to map the current economic value of the recreational services provided by European coastal ecosystems. For this purpose an extensive dataset of valuation studies is created and a series of explanatory variables such as biodiversity richness and geo-climatic variables are included in the model. These variables are selected in order to get a better and more economically oriented explanation of observed differences in ecosystem valuations. After inferring the main determinants of the willingness to pay (WTP) per person per year for cultural ecosystem services, values are aggregated to the entire relevant population of recreationists in order to estimate the current recreational value of coastal ecosystems in European countries.

1. The dataset of coastal and marine ecosystems values

A large data set of non-market valuations of coastal and marine ecosystems was constructed for the purposes of this study. In total, 320 primary valuation studies were retrieved and investigated. Online valuation databases constituted an important source of primary valuation studies or references to relevant papers. The investigation was not limited to the analysis of publications in the official scientific literature, but also explored “grey literature” (such as reports for both public and private institutions, consultancy studies, and unpublished working papers). Only primary valuations were considered and care was taken not to include more than once in the data set estimates that were published in multiple papers.

Of all studies, 79 were found to contain sufficient information for the meta-analysis. The total number of observations in the data set is 315. The average number of observations per study is 4.0 and the maximum number of observations per study is 24 (Downing & Ozuna 1996). The number of studies and observations is considerably larger than in previous meta-analyses of coastal and marine ecosystem values. A study on recreational value of coral reefs (Brander et al. 2007) counted 52 studies and 73 usable observations. A meta-analysis of contingent valuation studies in coastal and near-marine ecosystems (Liu & Stern 2008) collected 39 studies and 120 observations. Figure 2.14 presents the geographical distribution of the ecosystems in the dataset. Although all the continents are represented in the dataset, by far the largest number of observations (197 observations) is from the United States.

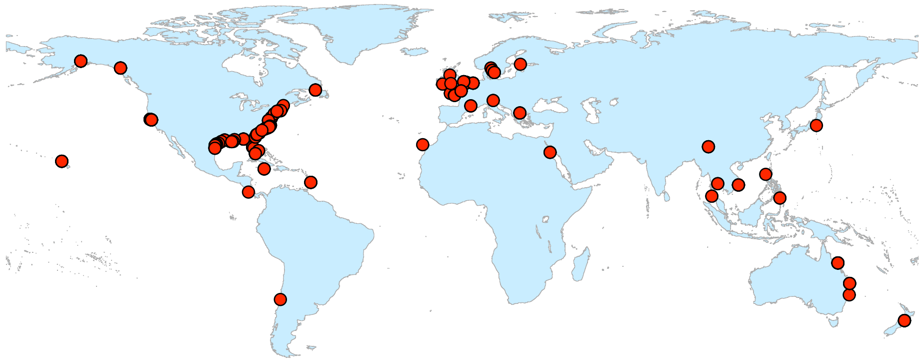

Figure 2.14. Geographical location of valued coastal ecosystems

All studies considered are primary valuation studies and no observation based on value transfer is included in the data set. In order to limit the risk of introducing publication bias, the investigation is not limited to the analysis of publications in the "official scientific literature", but also explores the "grey literature" (such as reports for both public and private institutions and consultancy studies) and unpublished research results. Efforts were also made to include studies that are not published in the English language. The average number of observations per study (4.0) and the maximum number of observations for a single study (24) are relatively low when compared to the total number of observations used in the analysis (315). As such, multiple sampling bias is expected to have a limited influence on the results of the investigation.

A large number of valuation studies focused on recreation, protection from erosion, and reduction of tourists congestion in sandy beaches (n=147). A number of observations are also available for conservation of biodiversity hotspots and recreation in coral reefs areas (n=25). A significant fraction of the total observations focused on marine and coastal protected areas (n=86). The largest valued ecosystem in the data set in terms of length of coastline is the Great Barrier Reef in Australia (Carr & Mendelsohn 2003) but smaller sites such as single beaches are also represented in the dataset.

Due to the focus on non-market values, the valuation methods included in the data set are either revealed or stated preference ones. Among the former, the contingent valuation method provided the largest number of observations (n=137), but choice experiments are also represented (n=18). The observations that were obtained with the travel cost method are 128. Finally, 32 values were estimated with the contingent behavior method, which combines both revealed and stated preference methods.

2. Specification of the meta-regression model

A semi-logarithmic model specification is assumed for the regression of the willingness to pay (WTP) per person per year for recreational activities in the valued sites. The final set of explanatory variables of the value function is chosen based on the experience gathered from previous meta-analyses of ecosystem values (Ghermandi et al. 2008). The explanatory variables are classified into three principle categories: valuation study characteristics, site characteristics, and context characteristics.

Study variables

The study characteristics that are accounted for in the meta-analytical value function are the type of valuation methods used in the primary study, the type of welfare measure elicited, and a dummy distinguishing whether values in the primary study are estimated for individuals or households. Valuation methods are classified into two categories according to the distinction between stated and revealed preference methods. Observations derived with stated preference method include contingent valuation, choice experiment, and contingent behaviour estimates. Revealed preference estimated by means of the travel cost method is the omitted category for valuation methods in the meta-regression. The type of welfare measure elicited in the primary valuation study is accounted for in the model by a dummy variable which reflects whether the observation is (i) a total WTP or total consumer surplus, or (ii) the WTP to achieve an increase (to forego a decrease) in the level of provision of a specific ecosystem service as compensating variation (equivalent variation). Finally a dummy is included to distinguish between values estimated for individuals and households since most primary valuation studies do not provide information on the average household size.

Site variables

The site characteristics that are accounted for in the meta-analytical value function are the type of ecosystem and the type of ecosystem service provided. A series of dummy variables is included in the model to distinguish between recreational activities that take place in sandy beaches (n=147), coral reefs (n=25) and other types of coastal ecosystems (n=143) (for example the latter include coastal marshes, rocky coastlines, and coastal forests).

Two types of cultural recreational ecosystem services are considered, i.e., recreational fishing (n=164) and non-consumptive recreation (n=206). The latter identifies activities such as swimming, snorkeling, diving, bathing, boating and beach leisure. Since the two types of services are not mutually exclusive, i.e., one value observation may reflect a combination of both, no reference category is defined in the analysis of this variable.

Context variables

Building upon the results of previous meta-analyses, context variables are introduced to capture the possible influence of income effects, population density, richness in biodiversity, and geo-climatic variables on ecosystem values. All variables are evaluated at a country level, with the exception of GDP per capita which is calculated at the state and NUTS2 levels for the United States and EU countries respectively. GDP per capita values were expressed in Purchase Power Parity (PPP) units and standardized to US$ 2003 per year following the procedure described in Ghermandi et al. (2008). The population density at country level is included in the meta-regression model in order to capture the fact that a high population density around the valued sites may contribute to the transformation of potential values into actual benefits as previously observed by Brander et al. 2006 and Ghermandi et al. (2008).

A series of geo-climatic and biodiversity variables were considered for their possible influences on the estimated values. The biodiversity variables considered include the total number of bird and mammal species and of threatened bird and mammal species at country level as derived from the Little Green Data Book (World Bank, The 2007). The geo-climatic variables evaluated for inclusion in the model include the average, minimum and maximum monthly temperature and the average yearly precipitation at country level. To reduce the negative effects of correlation between variables in the meta-regression, for the final set of variables we selected from this group the total number of bird species, threatened bird species, minimum and maximum monthly temperature only.

3. Econometric results

The estimated coefficients of the explanatory variables are mostly statistically significant and with the expected sign. Among study characteristics, the results of the regression confirm the *a priori* expectations that the value attributed to marginal changes in the level of provision of recreational services is lower than its total value, and that values expressing the WTP of a household are statistically higher than those referring to single individuals. No significant difference is found between stated and revealed preference valuation studies.

With respect to site characteristics, the coefficient for recreational activities in coral reefs is higher than in sandy beaches or in other coastal ecosystems, but the difference is not statistically significant. The value attributed by individuals to the recreational fishing experience is higher than that of non-consumptive recreational activities.

The coefficient of the variable ‘Real GDP per capita’ indicates the presence of income effects (Ghermandi et al. 2008; Brander et al. 2006). The coefficient of the population density variable is statistically significant but has a negative sign, which is in contrast with the findings of previous meta-analyses of the values of ecosystem services (Ghermandi et al. 2008; Brander et al. 2006). Both biodiversity variables have the expected signs, which indicate a higher value for recreational sites with high biodiversity and lower values where biodiversity is threatened. Finally, both coefficients on the temperature variables are positive and suggest that coastal recreational activities are more highly valued in warm climates.

4. Scaling up and distribution of values

The value of WTP per person per year for coastal recreational activities for 22 European countries during the baseline year 2003 was estimated based on the regression coefficients reported in Table 2. The calculated values reflect the total WTP for the provision of recreational services, i.e., the coefficient of the variable ‘variation’ is set equal to zero, and individual values, i.e., coefficient of ‘household’ equal to zero. The coefficients of the study method and the type of recreational activity reflect the sample average for the 315 observations. Values are calculated for a generic coastal ecosystem, i.e., the coefficients of both ‘beach’ and ‘reef’ are equal to zero. The value of the context variables during the baseline year is illustrated in Table 2.12.

Table 2.12. Values of context variables in baseline year regression (2003)

Country	Population density, inhab./km^2	GDP per capita, US$	Number of birds species	Number of threatened bird species	Min monthly temperature, °C	Max monthly temperature, °C
Belgium	340.81	32,808	427	10	2.12	16.94
Bulgaria	69.78	9,354	379	11	-3.14	21.82
Croatia	80.49	13,342	365	9	-2.45	19.90
Denmark	125.70	34,669	427	10	-1.32	15.13
Estonia	29.71	16,127	267	3	-15.36	15.20
Finland	15.59	32,678	421	10	-15.47	12.09
France	111.49	29,276	517	15	3.17	17.56
Germany	231.50	29,550	487	14	-1.53	17.29
Greece	84.13	24,399	412	14	12.90	24.11
Ireland	58.95	41,492	408	8	2.05	13.97
Italy	194.69	29,502	478	15	10.06	20.89
Latvia	35.64	13,540	325	8	-15.36	15.20
Lithuania	52.45	14,569	227	4	-9.11	15.91
Netherlands	393.20	33,198	444	11	5.99	14.58
Norway	14.31	41,630	442	6	-10.19	11.14
Poland	122.15	13,741	424	12	-5.22	17.58
Portugal	114.97	21,791	501	15	5.52	20.38
Romania	90.73	9,056	365	13	3.68	15.05
Slovenia	98.62	22,261	350	7	-2.45	19.90
Spain	85.96	26,296	515	20	3.69	20.87
Sweden	20.09	32,325	457	9	-0.40	4.37
UK	246.08	33,314	557	10	2.07	14.75

Table 2.13 presents the mean baseline values of WTP per person per year and the aggregated values of coastal recreational activities in the 22 European countries investigated. The total number of visitors per year represents the total number of domestic and international tourist arrivals in coastal NUTS2 regions in each of the considered countries, as estimated by Eurostat (2010) for year 2003.

Table 2.13. Aggregated WTP for coastal and marine recreation in Europe

Country	Average individual WTP [US$ /person year]	Arrivals in coastal NUTS2 (thousands) [a]	Aggregated value [Million US$/year]
Belgium	159.24	1,691	269
Bulgaria	103.58	1,330	138
Croatia	127.07	3,466	440
Denmark	144.73	2,951	427
Estonia	120.48	1,315	158
Finland	74.48	6,256	466
France	172.19	37,298	6422
Germany	110.12	9,385	1033
Greece	447.54	12,019	5379
Ireland [b]	250.55	8264	2071
Italy	282.54	48,662	13749
Latvia	49.70	659	33
Lithuania	104.54	560	59
Netherlands	153.51	9,199	1412

Norway	183.37	9,437	1730
Poland	71.40	975	70
Portugal	204.88	9,619	1971
Romania	70.56	907	64
Slovenia	195.08	1,015	198
Spain	176.64	47,383	8370
Sweden	110.62	12,911	1428
UK	178.24	39,334	7011

[a] Source: Total arrivals of residents and tourists according to Eurostat (2010) and referring to year 2003; [b] Number of arrivals refers to year 2000.

The highest WTP per person per year is found in the Mediterranean countries, Greece and Italy in particular. This is partly due to the fact that both minimum and maximum yearly temperatures are observed to be positively correlated with the values of WTP per person per year. WTP for tourism in Ireland and Norway is also high in spite of the low temperatures with respect to Mediterranean countries. This suggests that a different type of tourism may take place there, where climatic conditions are less crucial and tourists may be willing to pay more in order to enjoy the values of the natural landscape in a more pristine and less densely populated environment. Finland and Sweden have the lowest values of WTP per person per year, which suggests that in these countries the cold climate plays a crucial role in determining tourist demand.

Table 2.13 also provides estimates of the aggregated WTP values for all yearly visitors in the coastal regions of each considered country. High aggregated values are found in Mediterranean countries due to the fact that the estimated individual WTP in those countries is high and the tourism industry particularly developed there. High values are found in particular in Italy, France, and Spain. One of the highest total recreational values is found in the United Kingdom due to the high number of domestic and international arrivals reported for the reference year.

To spatially disaggregate the total values at country level, Figure 2.15 presents the values estimated for coastal NUTS2 regions in Europe. The values are obtained multiplying the individual WTP estimated at country level by the total number of arrivals in each region during year 2003, as reported by Eurostat.

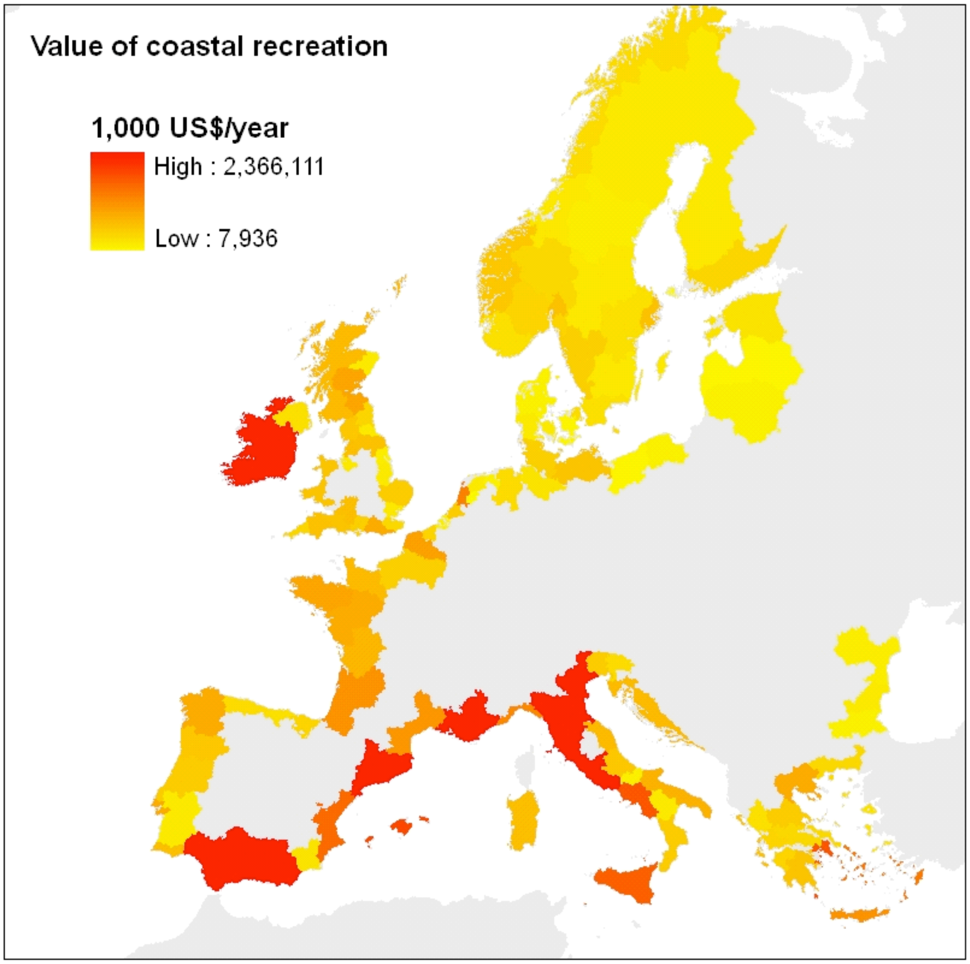

Figure 2.15. Aggregated values of recreation in coastal NUTS2 regions of Europe

The NUTS2 regions with the highest recreation values are those located in the Mediterranean coast of Italy, Spain, and France. Relatively high values are found also in the western coast of France and in the United Kingdom. A high aggregated value is found also for Ireland, although one should notice that due to lack of data relative to the number of visitors in NUTS2 regions of Ireland, it was not possible to disentangle the total country value into smaller units. Low aggregated recreation values are found in the Baltic and Scandinavia countries, and along the Black Sea coastline.

4. A GIS approach to investate the dependency of human livelihoods on bnefits of biodiversity and eocystem esrvices

To investigate the links and dependencies between a country's economy, biodiversity richness and the values of ecosystem services, we use a GIS to integrate spatially the three layers of information. The investigation undertaken in this section aims at testing the following hypotheses:

(1) A survey in the literature shows that environmental income can play a crucial role in the livelihoods of rural households, especially the poorest, therefore, poorer economies in Europe are expected to have stronger dependency on the ecosystem services.
(2) Despite biodiversity and environmental conservation policies are much more advocated in developed economies, larger proportion of the nature resources remaining pristine might be found in less developed economies as the resources are less extensively extracted for economic activities.
(3) The weights of each value component over the total economic value of ecosystem services differ across countries. Many factors may have influence on the weights of different values of ecosystem services in a country, however, the relative weight of provisioning services value might be expected to be higher in less advanced economies.
(4) Higher densely populated rural area might be found in coincidence with environmental resources concentrated area, e.g. high forest coverage; whereas the expansion of urban area is expected to be negatively correlated to the natural resources.

With these goals in mind, we selected the following indicators for the spatial integration:
- Biodiversity layer: index by Wendland et al. (2009), averaged at country level
- Socio-economic layer:
    - GDP per capita in 2007
    - Added value of agriculture as percentage of GDP in 2005
- Ecosystem services layer:
    - Total value and value per capita of the ecosystem services of forest, freshwater and coastal ecosystems considered in Section 10.

Table 2.14 presents the value of the selected indicators for the 28[11] European countries investigated.

Table 2.14. Indicators of socio-economic condition, biodiversity richness and ecosystem services value

Country	Ecosystem service value per capita (US$/person/year)	Added value of agriculture (% of GDP)	Biodiversity index	GDP per capita (US$, 2007, PPP)	Aggregated value [Million US$2003/year]
Austria	1,482	2	76.516	44,879	1,719
Belgium	789	1	62.127	42,609	2,805
Bulgaria	1,254	9	91.306	5,163	7,782
Croatia	3,477	7	76.896	11,559	11,914
Czech Republic	571	3	76.601	16,934	885
Denmark	441	1	36.503	57,051	1,020
Estonia	1,948	5	54.846	15,578	478

[11] Four countries are dropped from the dataset due to missing information.

Finland	5,204	3	39.695	46,261	9,599
France	442	2	76.456	41,970	4,344
Germany	417	1	68.987	40,324	7,739
Greece	1,513	4	62.744	27,995	10,847
Hungary	493	4	84.623	13,766	1,642
Ireland	3,527	2	22.934	59,324	11,640
Italy	1,212	2	67.138	35,396	46,862
Latvia	1,560	4	60.332	11,930	654
Lithuania	620	5	67.323	11,356	326
Luxembourg	876	0	71.935	103,042	89
Netherlands	641	2	49.158	46,750	4,687
Norway	2,504	2	29.654	82,480	3,692
Poland	461	5	70.825	11,072	3,423
Portugal	1,715	3	68.754	20,998	15,296
Romania	568	10	78.364	7,703	3,071
Slovakia	760	4	83.665	13,891	387
Slovenia	1,299	3	85.710	23,379	310
Spain	1,286	3	70.813	32,017	40,157
Sweden	7,132	1	34.143	49,662	38,658
Switzerland	616	1	77.355	56,207	1,027
United Kingdom	315	1	34.730	45,442	6,596

To analyze the relationships between indicators from each of the three layers considered (i.e., biodiversity, ecosystem services values and socio-economic characteristics) we selected different combinations of indicators and used two different approaches to clearly visualize the relationships between them across Europe.

First, using a GIS software we combined the indicators from the three layers in a mosaic map of Europe in which the color of each country represents a different combination of the indicators. To create the map we attribute to each indicator a band of color (green for biodiversity, red for ecosystem services values, blue for socio-economic indicators) and an intensity of color which changes from country to country reflecting the local value of the indicator. The intensity of the color ranges from 0 in the country with the lowest value to 255 in the country with the highest value. For instance, Bulgaria and Luxembourg have, respectively, the lowest and highest GDP per capita among the considered countries. Since the color blue is attributed to the socio-economic indicator, the intensity of blue in Bulgaria and Luxembourg will be respectively 0 and 255.

As a second approach to the visualization of the data, we produced radar charts for selected countries, in which each of the three axes represents one indicator.

A number of possible combinations of the different indicators can be investigated. In the following Figures, two ways of combining various indicators are demonstrated. Figures 2.16-17 combine the value of ecosystem services per capita, the biodiversity index and the GDP per capita at country level in Europe. Figures 2.18-19 illustrate the combination of the total value of ecosystem services as percentage of GDP, the biodiversity index, and the added value of agriculture as percentage of GDP.

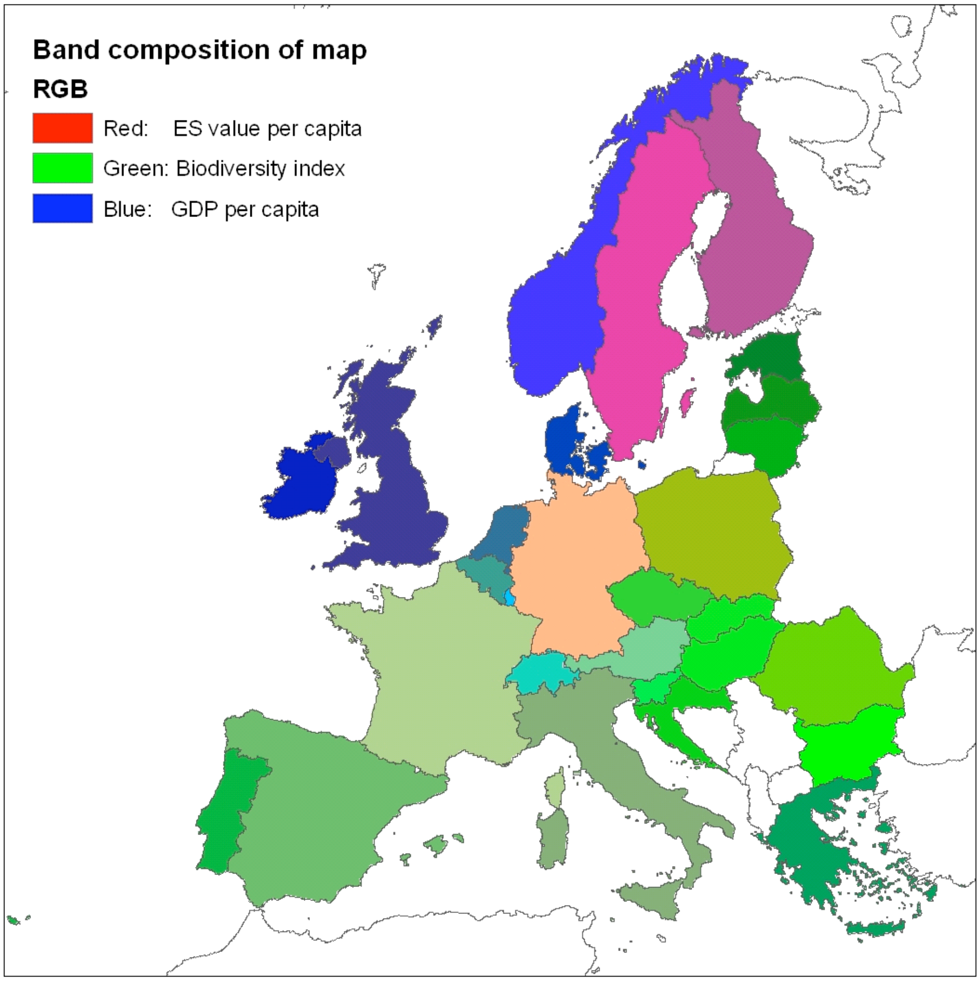

Figure 2.16. Mapping ecosystem services value, biodiversity index and GDP per capita.

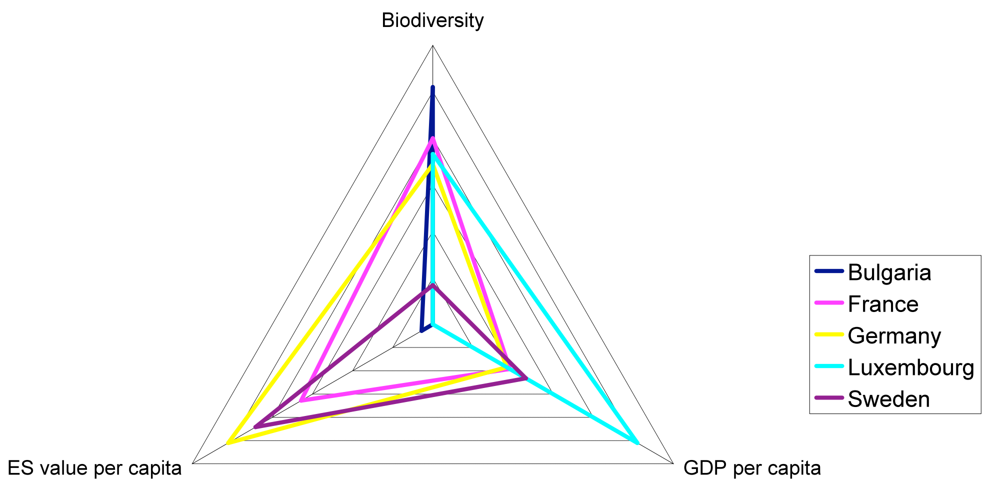

Figure 2.17 Radar chart of ecosystem services value, biodiversity index and GDP per capita for selected countries

The results show some interesting insights to the relationships between biodiversity, ecosystem benefits and human livelihoods. First, high values of culture services are found in the Mediterranean Europe (e.g. Italy, Spain, and France) for both coastal and forest ecosystems, both of which have also shown high values of biodiversity index. Second, in many of the eastern European countries, such as Poland, Romania and Czech republic, which are associated with low incomes but high value of biodiversity index, the values of ecosystem services are found very high, meaning that ecosystem service value may have an essential role in contributing to the local livelihoods in these countries. However exceptions are observed in Scandinavian countries, in particular Sweden and Finland, where both high values are found in terms of ecosystem service value per hectare and GDP per capita. This result may demonstrate that these two countries' economies largely depend on the ecosystem services, such as forest products, due to the large sizes of forest area for timber production and exportation.

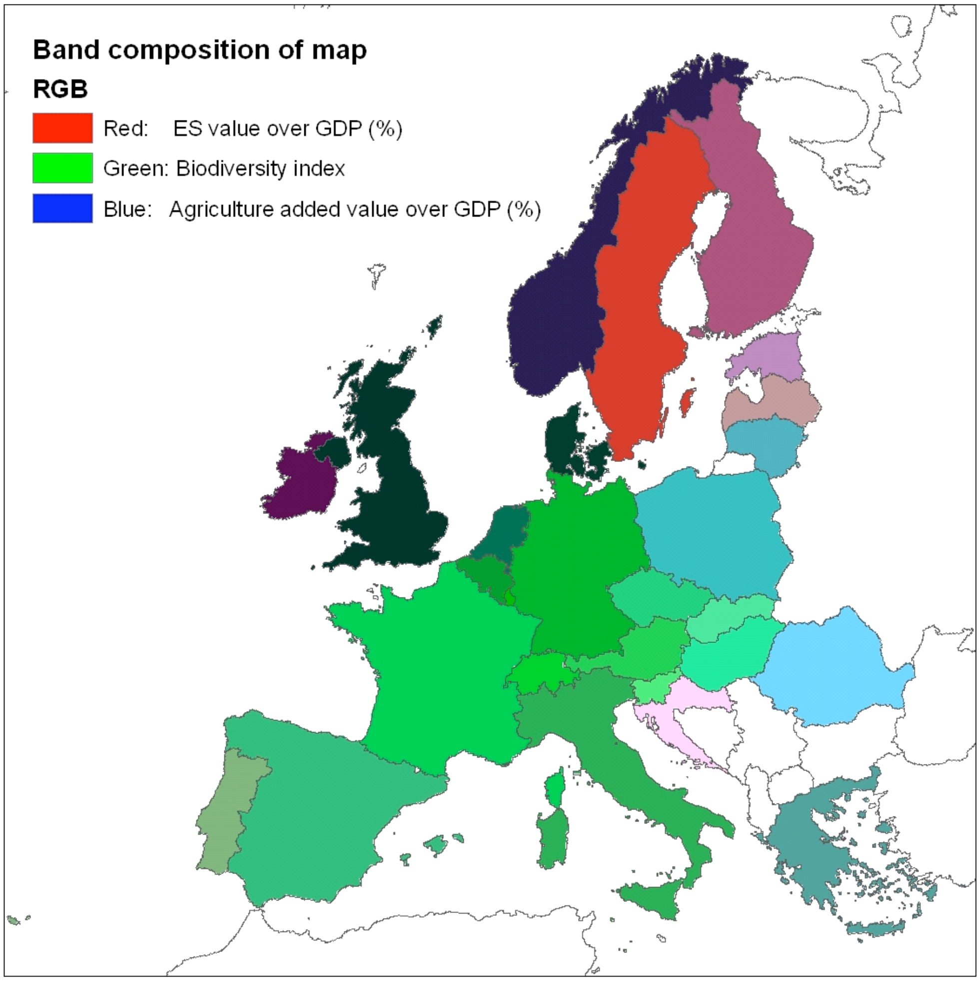

Figure 2.18 Mapping the weighed ES values over GDP, biodiversity index and added value of agriculture

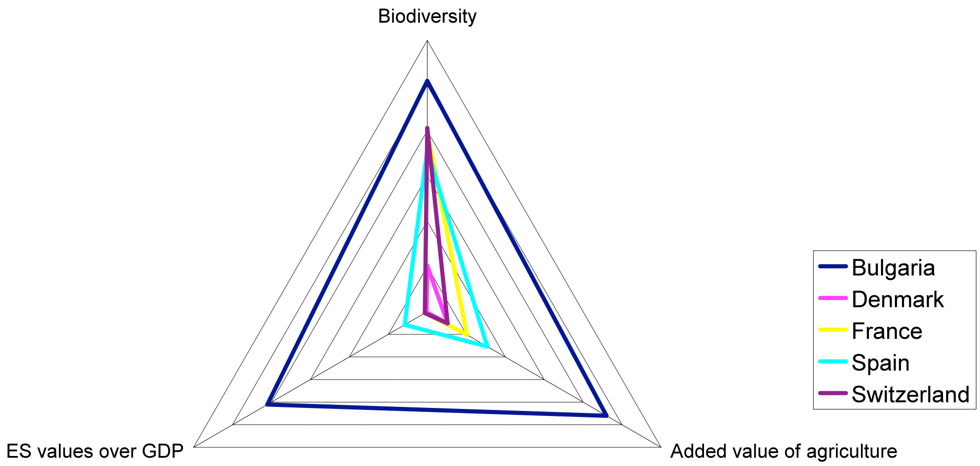

Figure 2.19. Radar chart of weighed ES values over GDP, biodiversity index and added value of agriculture

Figure 2.18-19 show that Eastern European countries, in particular Bulgaria, Croatia and Romania score high in all three indicators. This means these three countries are rich in the biodiversity resources, which also contribute substantially to the national economy. In other words, we may state that these countries' economies tend to have strong dependency on their ecosystem services. Similar findings are also hold for the three Baltic countries, i.e. Estonia, Latvia and Lithuania. As for the Northern Europe, several countries, e.g. Denmark, Ireland, UK and Norway score low for all three indicators. Note that ES value over GDP is measured in relative term, thus the absolute ES value can be still higher than the eastern European countries. However these relative low values reflect the fact that these countries' economies are established on more value added industries rather than ecosystem dependent economic sectors. Moreover, it is interesting to notice that some other countries have high biodiversity index but low ecosystem services values compared to the GDP. For instance, in the case of Switzerland and Luxembourg, it is most likely a result of the very high GDP per capita in these countries. As For other countries such as France and Spain we could argue that this might be due to the fact that provisioning services (for instance provided by forests) are relatively low (in terms of % of GDP). Despite the higher cultural values, the percentage of ecosystem services values over the GDP remains relatively low.

References:

Amigues, J.P. & Desaigues, B., 1999. L'évaluation d'une politique de protection de la biodiversité des forêts riveraines de la Garonne. La valeur économique des hydrosystèmes. Méthodes et modèles d'évaluation des services délivrés, Paris, Economica, 37-62.

Australian Government (2004) "Australian Greenhouse Office – Annual Report 2003-2004)"

Babulo, B., Muys, B. Nega, F., Tollens, E., Nyssen, J., Deckers, J., E. Mathijs, 2008. Household livelihood strategies and forest dependence in the highlands of Tigray, Northern Ethiopia. Agriculture Systems (98): 147-155.
Barbier, E.B. (1994) “Valuing Environmental Functions: Tropical Wetlands”, Land Economics, 70(2), pp. 155-173
Berrittella, M., Bigano, A., Roson, R. and R.S.J. Tol (2006), “A general equilibrium analysis of climate change impacts on tourism”, Tourism Management vol 25.
Bigano, A., Bosello, F., Roson, R. and R.S.J. Tol (2008) “Economy-wide impacts of climate change: a joint analysis for sea level rise and tourism", Mitigation and Adaptation Strategies for Global Change, Vol. 13, n. 8.
Bigano, A., Hamilton, J.M., & Tol, R.S.J., 2007. The impact of climate change on domestic and international tourism: A simulation study. Integrated Assessment Journal, 7(1), 25-49.
Bin, O. et al., 2005. Some Consumer Surplus Estimates for North Carolina Beaches. Marine Resource Economics, 20(2), 145.
Birol, E., Karousakis, K. & Koundouri, P., 2006. Using a choice experiment to account for preference heterogeneity in wetland attributes: the case of Cheimaditida wetland in Greece. Ecological Economics, 60(1), 145-156.
Bolin B., Sukumar R. et al., 2000. Global perspective. Land use, land-use change, and forestry. In: Watson, R. T. et al. (Eds.). A Special Report of the IPCC. Cambridge University Press, 23-51.
Bonnieux, F. & Le Goff, P., 1997. Valuing the benefits of landscape restoration: a case study of the Cotentin in Lower-Normandy, France. Journal of environmental management, 50(3), 321-333.
Bos, E.J. & Bergh, J., 1998. Economic Evaluation, Land/Water Use, and Sustainable Nature: Conservation of “De Vechtstreek” Wetlands. Dept. of Spatial Economics, Vrije Universiteit, Amsterdam.
Braden, J. B. and C. D. Kolstad (eds) (1991) “Measuring the Demand for Environment Quality”, Elsevier Science Publishers, North-Holland.
Brander, L. et al., 2008. Scaling up ecosystem services values: methodology, applicability and a case study, Eni Enrico Mattei Foundation (FEEM) and Institute for Environmental Studies VU University Amsterdam (IVM).
Brander, L., Florax, R.J.G.M. & Vermaat, J.E., 2006. The empirics of wetland valuation: A comprehensive summary and a meta-analysis of the literature. Environmental and Resource Economics, 33(2), 223-250.
Brander, L., van Beukering, P. & Cesar, H., 2007. The recreational value of coral reefs: A meta-analysis. Ecological Economics, 63(1), 209-218.
Brouwer, R., 2000. Environmental value transfer: state of the art and future prospects. Ecological Economics, 32(1), 137-152.
Bureau d'Etudes ACSA, 1996. Evaluation économique des services rendus par les zones humides. Study for Eaufrance. Available at: http://www.economie.eaufrance.fr/base_dommages/detail.php3?id_etude=8.
Büttner, G. et al., 2002. Corine land cover update 2000. European Environmental Agency, Copenhagen, Denmark.
Byström, O., 2000. The replacement value of wetlands in Sweden. Environmental and Resource Economics, 16(4), 347-362.
Carr, L. & Mendelsohn, R., 2003. Valuing coral reefs: a travel cost analysis of the Great Barrier Reef. Ambio, 32(5), 353-357.
Carson, R. T., L. Wilks and D. Imber (1994) “Valuing the Preservation of Australia's Kakadu Conservation Zone”, Oxford Economic Papers, Oxford, 46(5), pp. 727-749.
Clark, J (2001) “the global wood market, prices and plantation investment: an examination drawing on the Australian experience”, Environmental Conservation 28 (1): 53-64
Cline, W.R. (1992) “The Economics of Global Warming”, Institute for International Economics, Washington, DC.
Costanza Robert, d'Arge Ralph, de Groot Rudolf, Farber Stephen, Grasso Monica, Hannon Bruce, Limburg Karin, Naeem Shahid, O'Neill Robert V., Paruelo Jose, Raskin Robert G.,

Sutton Paul, van den Belt Marjan (1997) “The value of the world's ecosystem services and natural capital”, Ecological Economics, 25, pp. 3-15.
Cravener, M., 1995. Samhällsekonomisk värdering av den anlagda våtmarken i Oxelösund, en tillämpning av Contingent Valuation metoden. Master thesis. Department of Economics, Stockholm University, Sweden.
Dabat, M.H. & Rudloff, M.A., 1999. La valeur de préservation d'une lagune méditerranéenne menacée de comblement. La valeur économique des hydrosystèmes. Méthodes et modèles d'évaluation des services délivrés. Economica, Paris, 107-135.
de Groot, A.W.M. & Velthuijsen, J.W., 1998. Natuurlijk vermogen: een empirische studie naar de economische waardering van natuurgebieden in het algemeen en de Oostvaardersplassen in het bijzonder, SEO, Stichting voor Economisch Onderzoek der Universiteit van Amsterdam.
DEFRA (2007) “The Social Cost of Carbon and The Shadow Price of Carbon: What They Are, and How to Use Then in Economic Appraisal in the UK”, Economics Group of the Department for Environment, Food and Rural Affairs, UK. Available online at:
Depondt, F. & Green, E., 2006. Diving user fees and the financial sustainability of marine protected areas: Opportunities and impediments. Ocean & Coastal Management, 49(3-4), 188-202.
Desaigues, B., 1991. Bénéfices écologiques et récréatifs du lac de la Forêt d'Orient. Revue d'Economie Politique.
Downing, M. & Ozuna, J., 1996. Testing the Reliability of the Benefit Function Transfer Approach. Journal of Environmental Economics and Management, 30(3), 316-322.
El Yousfi, H., Nicolai, S. & Casin, P., 2006. Etude économique sur les coûts et bénéfices environnementaux dans le domaine de l'eau : l'île de Rhinau. Agence de l'Eau Rhin-Meuse, Université Paul Verlaine de Metz, Rapport de stage.
FAO/FRA 2005 (2006), Global Forest Resources Assessment 2005: Progress towards sustainable forest management, FAO Forestry Paper no.147, available on website: ftp://ftp.fao.org/docrep/fao/008/A0400E/A0400E00.pdf
Folke, C., 1991. The societal value of wetland life-support. Linking the Natural Environment and the Economy: Essays from the Eco-Eco Group, 141.
Freeman III, A.M., 1995. The Benefits of Water Quality Improvements for Marine Recreation: A Review of the Empirical Evidence. Marine Resource Economics, 10(4), 385-406.
Garcia-Gonzalo J., Peltoa H., Gerendiain A. Z., Kellomäki S., 2007. Impacts of forest landscape structure and management on timber production and carbon stocks in the boreal forest ecosystem under changing climate, Forest Ecology and Management, 241: 243-257.
Garrod, G.D., Willis, K.G. & Saunders, C.M., 1994. The benefits and costs of the Somerset Levels and Moors ESA. Journal of rural studies, 10(2), 131-145.
GBA (1995) Global Biodiversity Assessment, published for the United Nation Environment Programme, Cambridge University Press
Georgiou, S. et al., 1998. Determinants of individuals' willingness to pay for perceived reductions in environmental health risks: A case study of bathing water quality. Environment and Planning A, 30, 577-594.
Ghermandi, A. et al., 2008. The economic value of wetland conservation and creation: A meta-analysis. Fondazione Eni Enrico Mattei Working Papers, 238.
Giardina, C.P., and M.G., Ryan (2000), ‘Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature’, Nature 404: 858-861
Gosselink, J.G., Odum, E. & Pope, R., 1974. The value of the tidal marsh. Center for Wetland Resources, Louisiana State University, Baton Rouge, LSU-SG-74-03.
Gren, I. & Soderqvist, T., 1994. Economic value of wetlands: a survey. Beijer Discussion Paper Series, 54.
Gren, I.M., Groth, K.H. & Sylvén, M., 1995. Economic values of Danube floodplains. Journal of Environmental Management, 45(4), 333-345.
Hammitt, J.K. (2000). Valuing mortality risk: theory and practice. Environmental Science and Technology 34: 1394–1400.
Hanley, N. & Craig, S., 1991. Wilderness development decisions and the Krutilla-Fisher model: the case of Scotland's' flow country', Department of Economics, University of Stirling.

Hanusch, H., Cantner, U. & Muench, K., 2000. Erfassung und Bewertung der Umweltwirkungen des Ausbaus der Donaustrecke Straubing - Vilshofen. Available at: http://www.donauforum.de/article_detail.php? site_id=4& article_id=36.
Hein, L. et al., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecological economics, 57(2), 209-228.
Heston, A., Summers, R. & Aten, B., 2006. Penn world table version 6.2. Center for International Comparisons of Production, Income and Prices at the University of Pennsylvania.
http://www.opsi.uk/click-use/value-added-licence-information/index.htm
IEEP – Institute for European Environmental Policy (2006) "Value of Biodiversity: Documenting EU Examples Where Biodiversity Loss Has Led to the Loss of Ecosystem Services", ENV.G.1/FRA/2004/0081, Final Report
IMAGE (2001) Integrated Model to Assess the Global Environment, Netherlands Environmental Assessment Agency – RIVM, available at http://www.rivm.nl/image/
IPCC, 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
IPPC, 2001. Intergovernmental Panel on Climate Change, Climate Change 2001: The Scientific Basis. Cambridge University Press, Cambridge, UK, pp. 881
Jarvis, P.G. and S., Linder (2000), 'Botany-constraints to growth of boreal forests', Nature 405: 904-905
Júdez, L. et al., 1998. «Évaluation contingente de l'usage récréatif d'une réserve naturelle humide». Cahiers d'Économie et Sociologie Rurales, 48, 38-60.
Kamanga, P., Vedeld, P., E. Sjaastad, 2009. Forest Incomes and rural livelihoods in Chiradzulu District, Malawi. Ecological Economics 68: 613-624
Kosz, M., 1996. Valuing riverside wetlands: the case of the "Donau-Auen" national park. Ecological Economics, 16(2), 109-127.
Ledoux, L., 2003. Wetland valuation: state of the art and opportunities for further development. In Proceedings of a Workshop Organised for the Environment Agency by Environmental Futures Ltd. and CSERGE. Bristol: Environment Agency. pp. 5–17.
Lehner, B. & Döll, P., 2004. Development and validation of a global database of lakes, reservoirs and wetlands. Journal of Hydrology, 296(1-4), 1-22.
Leitch, J.A. & Hovde, B., 1996. Empirical valuation of prairie potholes: five case studies. Great Plains Research, 6(1), 25-39.
Liu, S. & Stern, D.I., 2008. A Meta-Analysis of Contingent Valuation Studies in Coastal and Near-Shore Marine Ecosystems. Available at: http://mpra.ub.uni-muenchen.de/11720/ [Accessed January 15, 2009].
Lloyd, J. and J.A. Taylor (1994), 'On the temperature dependence of soil respiration', Funct.Ecol.8: 315-323
Loomis, J., Yorizane, S. & Larson, D., 2000. Testing Significance of Multi-Destination and Multi-Purpose Trip Effects in a Travel Cost Method Demand Model for Whale Watching Trips. Agricultural and Resource Economics Review, 29(2), 183-191.
Luo, Y.Q., Wan, S.Q., Hui, D.F., Wallance, L.L. (2001), 'Acclimatization of soil respiration to warming in a tall grass prairie', Nature 413: 622-625
Mäler, K-B. (1988) "Production function approach in developing countries", in Valuing Environmental Benefits in Developing Countries, Vincent et al. (eds), Special Report, Michigan State University, pp. 11-32.
Marangon, F., Tempesta, T. & Visintin, F., 2002. Turismo e attività ricreative nelle aree protette italiane: un quadro conoscitivo ancora inadeguato. In Paper presented at 2nd Conferenza Nazionale delle Aree Naturali Protette. pp. 11–13.
Markandya, A., Chiabai, A., Ding, H., Nunes P.A.L.D and C. Travisi (2008) " Economic Valuation of Forest Ecosystem Services: Methodology and Monetary Estimates", in "Cost of Policy Inaction" Final report for the European Commission, Brussels, Belgium.
MCPFE (2007), State of Europe's forests 2007: The MCPFE report on sustainable forest management in Europe, Ministerial Conference on the Protection of Forests in Europe, Liaison Unit Warsaw, Poland. ISBN: 83-922396-8-7 or 978-83-922396-8-0

MCPFE and UNECE/FAO (2003) State of Europe's Forests 2003 - Report on Sustainable Forest Management in Europe, Vienna, Austria.
MEA (2003) "Ecosystems and Human Well-being: A Framework for Assessment", World Resources Institute, Washington, D.C.
MEA (2005) "Ecosystems and Human Well-being: Biodiversity Synthesis", World Resources Institute, Ashington, D.C.
Melillo, J.M., McGuire, A.D,, Kicklighter,D.W., Moore III, B., Vorosmarty, C.J., Schloss, A.L. (1993), 'Global climate change and terrestrial net primary production', Nature 363: 234-239
Merlo, M. and L. Croitoru (2005) Valuing Mediterranean Forests: Towards Total Economic Value, CABI Publishing-CAB International, Wallingford, Oxfordshire, UK.
Meyerhoff, J. & Dehnhardt, A., 2007. The European Water Framework Directive and economic valuation of wetlands: the restoration of floodplains along the River Elbe. European Environment, 17(1).
Millennium Ecosystem Assessment, 2005. Ecosystems and Human welfare: Synthesis. Island Press, Washington, DC.
Mitchell, R.C. and R.T. Carson (1989), Using Surveys to Value Public Goods: The Contingent Valuation Method (Washington, DC: Resources for the Future).
Nakicenovic N., Swart R., 2000. IPCC Special Report on Emission Scenarios (Cambridge University Press, Cambridge.
Navrud, S., 1993. Cost Benefit Analysis of Liming Lake Vegar (Samfunnsùkonomisk lùnnsomhet av a kalke Vegar). Trondheim: Directorate for Nature Management, Report (Utredning for DN) 1993-5, 52 pp. (In Norwegian).
NOAA - National Oceanic and Atmospheric Administration (1993) "Report of the NOAA Panel on Contingent Valuation", Federal Register, 58(10), pp. 4601-4614, US.
Nordhaus, W.D (1991a) "A Sketch of the Economics of the Greenhouse Effect", American Economic Review, Papers and Proceedings 81: 146-150.
Nordhaus, W.D (1991b) "To Show or Not to Slow: the Economics of the Greenhouse Effect", Economic Journal 101: 920-937.
Nordhaus, W.D (1993a) "Optimal Greenhouse Gas Reductions and Tax Policy in the 'DICE' model", American Economic Review, Papers and Proceedings 83: 313-317.
Nordhaus, W.D (1993b) "Rolling the 'DICE': an Optimal Transition Path for Controlling Greenhouse Gases", Resources and Energy Economics 15: 27-50.
Nunes, P.A.L.D., J.C.J.M. van den Bergh and P. Nijkamp (2003) "The Ecological Economics of Biodiversity: Methods and Policy Applications", Edward Elgar Publishing (UK).
OECD (1999) "Environmental Indicators for Agriculture: Volume 1 Concepts and Frameworks", Organization for Economic Co-operation and Development, Pairs.
Oglethorpe, D.R. & Miliadou, D., 2000. Economic Valuation of the Non-use Attributes of a Wetland: A Case-study for Lake Kerkini. Journal of Environmental Planning and Management, 43(6), 755-767.
Pearce, D. and D. Moran (1994) "The Economic Value of Biodiversity", Earthscan Publications Limited, London, UK.
Perrings, C.A., Mäler, D.-G., Folke, C., Holling, C.S. and B.-O. Jansson (1995) Biodiversity Conservation, Kluwer Academic Publishers, The Netherlands
Posford Duvivier Environment, 1999. River Ancholme flood storage area progression. Report no. E3475/01/001. Environment Agency, UK.
Posford Duvivier Environment, 2000. River Nar improvement scheme. Engineers report. Environment Agency, UK.
Ragkos, A. et al., 2006. Using a functional approach to wetland valuation: the case of Zazari–Cheimaditida. Regional Environmental Change, 6(4), 193-200.
Riou-Nivert P., 2005. Les temps changent, la forêt doit s'adapter, Forêt-entreprise, 162: 12.
Scherr, S.J., White, A., and D. Kaimowitz, 2003. A new agenda for forest conservation and poverty reduction: making forest markets work for low-income producers. Forest Trends. ISBN 0-9713606-6-9
Scherrer, S., 2003. Evaluation économique des aménités récréatives d'une zone humide intérieure: le cas du Lac du Der, MEDD, Direction des Etudes Economiques et de l'Evaluation Environnementale, document de travail.

Schöter D. et al., 2004. ATEAM (Advanced Terrestrial Ecosystem Analyses and Modelling) final report (Potsdam Institute for Climate Impact Research, 2004).
Schöter D., Cramer W., Leemans R., Prentice I. C., Araùjo M. B., Arnell N. W., Bondeau A., Bugmann H., Carter T. R., Gracia C. A.,. de la Vega-Leinert A. C, Erhard M., Ewert F., Glendining M., House J. I., Kankaanpää S., Klein R. J. T., Lavorel S., Lindner M., Metzger M. J., Meyer J., Mitchell T. D., Reginster I., Rounsevell M., Sabaté S., Sitch S., Smith B., Smith J., Smith P., Sykes M. T., Thonicke K., Thuiller W., Tuck G., Sönke Zaehle, Bärbel Z., 2005. Ecosystem Service Supply and Vulnerability to global change in Europe. Science, 310: 1333-1337.
Seguì-Amórtegui, L., 2004. Sistemas de regeneración y reutilización de aguas residuales. Metodología para el análisis técnico-económico y casos. Doctoral Thesis. Departamento de Ingeniería Agroalimentaria y Biotecnología. Universidad Politécnica de Cataluña.
Shrestha, K., Seidl, A. & Moraes, A., 2002. Value of recreational fishing in the Brazilian Pantanal: a travel cost analysis using count data models. Ecological Economics, 42(1-2), 289-299.
Tavoni, M. B. Sohngen, et al. (2007). “Forestry and Carbon Market Market Response to Stablize Climate”. Climate Change Modeling and Policy Working Papers. Fondazione Eni Enrico Mattei (FEEM):17.
Thuiller W., Lavorel S., Araùjo M. B., Sykes M. T., Prentice I. C., 2005. Climate change threats to plant diversity in Europe. PNAS, vol. 102, n° 23: 8245-8250.
Turner, R. & Brooke, J., 1988. Management and valuation of an environmentally sensitive area: Norfolk Broadland, England, case study. Environmental Management, 12(2), 193-207.
UNECE/FAO, 2005. European Forest Sector Outlook Study, 1960 – 2000 – 2020, Main report, United Nation Publications, pp. 265
UNEP-WCMC, 2007. Biodiversity and poverty reduction: the importance of biodiversity for ecosystem services. www.unep-wcmc.org
Walker B. H., Steffen W. L., Canadell J., IngramJ. S. I. (eds), 1999. The terrestrial biosphere and global change: implication for natural and managed ecosystems. Synthesis volume. IGBP Book Series 4, Cambridge University Press, pp. 450
WBCSD, 2009. Corporate ecosystem valuation: a scoping study. Prepared for the World Business Council for Sustainable Development by Environmental Management Group & Ecosystem Economics LLC.
Woodward, R.T. & Wui, Y.S., 2001. The economic value of wetland services: a meta-analysis. Ecological Economics, 37(2), 257-270.
World Bank, 2002. Linking poverty reduction and environmental management.
World Bank, 2006. World development indicators. World Bank Publications. Available at: http://devdata.worldbank.org/wdi2006/contents/index2.htm.
World Bank, The, 2007. Little Green Data Book.