ECONADAPT

The Economics of Adaptation

Funded by the European Union

# Future values for adaptation assessment

M. Skourtos, Damigos D., Kontogianni A., Tourkolias C., Markandya, A., Abadie, L. Mª, Sainz de Murieta, E., Galarraga, I., Wellman, J. and A. Hunt



| | |
|---|---|
| Deliverable number | D2.2 |
| Work Package Number | 2 |
| Submission date | December 2015 |
| Type of Activity | RTD |
| Nature | R |
| Dissemination level | Public |

# Document information

| | |
|---|---|
| Title: | Future values for adaptation assessment |
| Authors: | Skourtos, M., Damigos D, Kontogianni A., Tourkolias C., Markandya, A., Abadie, L. Mª, Sainz de Murieta, E., Galarraga, I., Wellman, J. and A. Hunt |
| Other Contributors | |
| Date: | February 2016 |
| Contact details | Michail Skourtos mskour@aua.gr |
| Work Package Number | 2 |
| Deliverable number | D2.2 |
| Filename: | D2.2_future preferences.docx |
| Document history: | Draft November 2015/ Final version February 2016 |
| Type of Activity | RTD |
| Nature | R |
| Dissemination / distribution level | PU |
| Citation: | Deliverable 2.2 of the ECONADAPT Project. |
| Copyright: | |

The ECONADAPT project has received funding from the European Union's Seventh Framework Programme for research, technological development and demonstration under grant agreement no 603906.

To find out more about the ECONADAPT project, please visit the web-site: www.econadapt.eu

For further information on the ECONADAPT project contact Alistair Hunt at: ecsasph@bath.ac.uk

# Table of Contents

# Sub-task 1: Developing methods to model changing preferences over future time periods.

# Executive Summary

Within the framework of ECONADAPT project Work Package 2 addresses key methodological questions in the microeconomic foundations of climate change adaptation assessment. The first part of the document at hand reports on research conducted within Task 2b: Future values for adaptation assessment, Sub-task 1. Developing methods to model changing preferences over future time periods. The aim of Deliverable D2.2 is to construct a robust methodology for investigating future paths of preferences and values related to economic adaptation assessment. The proposed methodology is illustratively implemented to estimate future values (up to 2050) for temperate and boreal forests that constitute a significant part of the total forest area in Europe. From the totality of ecosystem services provided by temperate and boreal forests our illustrative analysis focuses on the recreational opportunities. To this end, the parameters of the model (e.g. the WTP elasticity of demand) are selected accordingly.

The importance of this relates to present policies which often have impacts that extend into the far future. In that sense, present policies subtly impinge on the welfare of future generations (posterity) above and beyond that of present constituencies. Climate change is profoundly about the future; its impacts are going to be felt by future generations. The timing, spatial scale and degree of impacted assets and social groups are nevertheless beset with uncertainties. Key uncertainties surrounding the economic estimates of climate change damages include the evolution of markets and societies; the future growth of wealth and its distribution; changes in consumption modes and habits. To put it in a nutshell: uncertainty about future generations and their own preferences. The bulk of the discussion about future generations focuses on how much we should discount but there is not an adequate account until now of what to discount. The question then rises: who are future generations and how could their welfare be taken into consideration by present policies? The present report addresses this question.

Generally, preferences can be considered stable in the short-term future, but significant changes are expected in the long-term future. To this direction, strong indications have been identified that future generations will tend to be more sensitive about the environment (greening of preferences) leading to higher willingness to pay (WTP) values. Indisputably, the deterioration of the quality of environmental assets in combination with the upcoming depletion of natural resources constitutes the main reason for this tendency. The evolution of WTP values will be influenced mainly by the growth of income, depletion of environmental assets, elasticity of substitution between man-made and environmental goods and services and change in preferences of future generations. The formal expression for the growth rate of $WTP_t$ is therefore given by:

$$\alpha_{tot} = \alpha_{inc} + \alpha_{sc} + \alpha_{pr}$$

where $\alpha_{tot}$ is the total growth rate of WTP; $\alpha_{inc}$ the income growth factor; $\alpha_{sc}$ the environmental depletion (or scarcity) factor; $\alpha_{pr}$ the changing preferences factor. Since our focus is in the very long term, it is not possible to predict the magnitude and timing of future events and preferences that might influence the growth rate $\alpha_{tot}$. Therefore, simulation functions were constructed using flexible random walk-based stochastic models with and without drift (i.e. with and without change of preferences) to deal with the uncertainties involved.

The random walk-based stochastic process is equivalent to a Brownian motion. Based on the general model, the aggregate growth rate α of WTP in period (t) is estimated as follows:

$$\alpha_{tot}(t) = \theta(0) + k_t + \varepsilon_t$$

where: $\alpha_{tot}(t)$ is the total growth rate of WTP at time t; θ(0) the sum of $\alpha_{inc} + \alpha_{sc}$ at time 0; k the drift that reflects changes in future preferences; $\varepsilon_t$ a random component estimated by σW(t). Since the growth rate of WTP is a continuous process dependent on its rate of change and time, in order to properly characterize it we should express $\alpha_{tot}(t)$ as dependent on the differential change of the rate, i.e. it has to be rewritten as a differential process:

$$d\alpha_{tot}(t) = kd_t + \sigma dW(t)$$

For illustrative purposes, we apply the model to estimate the effect of growth rate $\alpha_{tot}$ on the WTP values of temperate and boreal forests that constitute a significant part of the total forest area in Europe. From the totality of ecosystem services provided by temperate and boreal forests our illustrative analysis focuses on the recreational opportunities. To this end, the parameters of the model are selected accordingly. Present WTP is estimated on the basis of the Ecosystem Service Value Database used in TEEB and presented by de Groot et al. (2012). In order to offset influences concerning differences of income, price level and time, we express the original values to Euros in 2015 prices using the methodology for benefit transfer proposed by Pattanayak et al. (2002). The rest of model parameters are estimated under a number of assumptions. The preferences factor $\alpha_{pr}$ is defined ad hoc under three different behavioural scenarios: Scenario A: Stable preferences; Scenario B: Green preferences; Scenario C: Materialistic preferences. In order to represent the wider range of uncertainty that affects future growth rate $\alpha_{tot}$, the Maximum Entropy approach was chosen. The idea behind maximum entropy is to formulate a distribution for the data such that the distribution maximizes the uncertainty in the data, subject to known constraints.

The main strength of the approach is its ability to handle the volatility of the parameters employing a Monte Carlo simulation for the total growth rate $\alpha_{tot}$. Its main weakness is its central assumption that prices are not influenced by past events. This idea stems from experience in financial markets where investors react instantaneously to any informational advantages they have thereby eliminating profit opportunities. This leads to a random walk where the more efficient the market, the more random the sequence of price changes. This is obviously problematic in the case of non-market assets - like ecosystem services – where no (efficient) markets exist.

Based on the results of the model estimation of future paths of WTP values of temperate and boreal forests the following remarks follow:

## Deterministic vs. probabilistic assessments

In all scenarios studied the probabilistic assessment results in higher estimates than the deterministic analysis. The disparities between the deterministic estimates and the probabilistic simulations are attributed primarily to the wide range of present WTP values.

## With and without the total growth rate $\alpha_{tot}$

The effect of total growth rate $\alpha_{tot}$ on the estimated PV of WTP is significant, even when preferences are assumed to remain constant. The effect of total growth rate $\alpha_{tot}$ is even more apparent when changes in the preferences of individuals are involved in the stochastic model.

## With and without changing preferences

The comparison of the estimates for the three Scenarios A, B and C reveals the importance of considering the effect of evolving preferences, especially in long-term analyses. In fact, the PV of WTP for constant preferences is about 35% higher than the estimated PV for the case of materialistic preferences and about 30% lower than the PV estimated for the case of green preferences.

The investigation of future preferences and changing values are a central object of inquiry referring to the estimation of climate change damages. The analytical encounter with the problem of future preferences is central to the economics of adaptation assessment since the economic rationale of investing in adaptation projects strongly hinges on the estimation of avoided future damages. Our random walk model allows the analyst to visualize future paths of preference and value evolution and by doing so bring future values of damaged assets realistically to the fore. For example, the comparison between the estimates of Scenarios B and C highlights the significance of the assumptions adopted regarding the evolution of preferences in the next decades: the PV of forests corresponding to greening of preferences is around 88.5% higher than the corresponding value of Scenario C. This finding is worrisome, considering that future preferences are unknown since complex and interlinked socioeconomic and behavioural factors are involved, which are also changeable. Therefore, potential behavioural patterns should be considered in the analyses, at least for sensitivity purposes.

The reliability of the model crucially depends on the reliability of input data referring to the elasticities of demand and income as well as projected growth rate of world economies. A step forward therefore is the embedment of our model into Shared Socioeconomic Pathways (SSPs); this would add to the completeness and reliability of parameter estimation and integrate the model into the wider discussion of socio-economic pathways for adaptation assessment.

# 1 Introduction

## 1.1 Task description, aims and objectives

The aim of the ECONADAPT project is to provide user-orientated methodologies and evidence relating to economic appraisal criteria to inform the choice of adaptation actions using analysis that incorporates cross-scale governance under conditions of uncertainty. The objective is to provide policymakers with the necessary framework and resources to design adaption policies by applying state-of-the-art analytical techniques to deeply uncertain and value-loaded issues. The main outcome of this research effort would improve the economic evidence base for adaptation. To this purpose, the microeconomic foundations of present modelling approaches, and especially Integrated Assessment Models (IAMs), need to be to investigated and improved.

The document at hand reports on research conducted within the frame of Task 2b: Future values for adaptation assessment, Sub-task 1. Developing methods to model changing preferences over future time periods. The Work Package description is outlined below.

This sub-task will undertake an informal meta-analysis of available data to identify and quantify parameters that can be interpreted as independent variables in the determination of key generic preferences in future time periods. Relevant parameters - including income – will be identified and extrapolated for future time periods to the 2050s. This will be undertaken in the context of the SSP socio-economic development pathways currently being developed by the IPCC. This extrapolation modelling would be extended by the development of simulation value functions constructed from a combination of observed relationships and understandings of value determination elicited from existing stated preference data elicited from household-based interviews, as outlined by Kilbourne et. al. (2005).

Thus, the goals of research in Task 2b, Subtask 1 are:

- The quantification of independent variables in the determination of key generic preferences in future time periods
- The extrapolation of these variables for future time periods to the 2050s
- The development of simulation value functions

While expected output includes:

- A robust methodology to investigate future paths of preferences and values relayed to economic adaptation assessment
- A first, illustrative implementation of the methodology to estimate future values for selected market/non market ecosystem services (up to 2050)

## 1.2 Framing the problem of future preferences and values

Not infrequently, present policies have impacts that extend into the far future (i.e. policies addressing issues in health, insurance, education, infrastructure, space exploration etc.). In that sense, present policies subtly impinge on the welfare of future generations (posterity) above and beyond that of present constituencies. Climate change is profoundly about the future; its impacts are going to be felt by future generations. Mitigation policies are accordingly impacting future generations only, meaning that present cost is counterbalanced with future benefits. On the other hand, adaptation policies benefit future generations but to a lesser degree also the present one. The timing, spatial scale and degree of impacted assets and social groups are nevertheless beset with uncertainties. The economic dimension of climate change uncertainties is better encapsulated in efforts to quantitatively estimate the social cost of carbon (SCC). SCC is the marginal external cost of a unit emission of CO2, expressed in terms of forgone consumption and based upon the damages inflicted by that emission upon global society through additional climate change. The value of the SCC is generally estimated in an integrated assessment modelling (IAM) framework that couples a baseline socio-economic scenario, a climate carbon cycle model and a function for transforming temperature change into economic damages (Kopp and Mignone 2012). U.S. Department of Energy (2010) (cited by Kopp and Mignone 2012) identified a number of limitations with the three IAMs (i.e. DICE 2007, PAGE 2002 and FUND 3.5) it employed to calculate climate change damages. The limitations of the IAMs manifestly target the main uncertainties surrounding the physical and socio-economic aspects of climate change. In particular, it is noted that all three models (Kopp and Mignone 2012):

- Incompletely treated non-catastrophic damages, for instance omitting ocean acidification and other effects on ecosystem services;
- Incompletely treated potential catastrophic damages, such as the effects of major reorganizations of ocean circulation or massive ice sheet melt;
- Crudely extrapolated damages calibrated at low degrees of warming (around 2.5C$^{o}$) to high degrees of warming (in some scenarios, 10C$^{o}$ or more);
- Failed to incorporate inter-sectoral interactions (such as the effects of water resources on agriculture) and inter-regional interactions (such as the effects of human migration between regions);
- Did not account for the imperfect substitutability of environmental amenities, assuming instead that it is possible to fully replace damaged natural systems with market goods; and
- Incompletely and opaquely treated adaptation to climate changes.

The cautious reader is puzzled by the list of limitations presented above: Where is the uncertainty surrounding the evolution of markets and societies; the future growth of wealth and its distribution; changes in consumption modes and habits? To put it in a nutshell: where are future generations and their own preferences? Despite a growing discussion about uncertainty inherent in climate change impact assessment most of the economic estimations of climate damages (and consequently adaptation benefits) are salient towards the preferences of future generations. Socio-economic scenarios (and SSPs for that) do not take explicitly into account future preferences. The bulk of the discussion about future generations focuses on how much we should discount but there is not an adequate account until now of what to discount! (Sterner and Persson 2008). The question then rises: who are future generations and how could their welfare be taken into consideration by present policies?

Present generations invest in climate change adaption and mitigation for two reasons: An ethical one meaning a moral sense of responsibility towards the yet unborn. And a rights aspects meaning that posterity has as much right as we did to inherit a hospitable Earth (Summers and Zeckhauser 2008). Whatever the case may be, our attitude towards posterity is either guided by our own preferences about 'good and evil' and therefore present patterns of relative social worth of man-made and ecosystem assets. In this case we exhibit a paternalistic altruism to future generations. Or, alternatively, we take future preferences seriously and conjecture as to what would future generations themselves consider a welfare-enhancing pattern of man-made and ecosystem assets. This is a non-paternalistic attitude to future generations (Horowitz 2002).

The lack of studies on future preferences in climate change mitigation and adaption assessment is the most curious as there are already similar efforts in areas such as: health care (Dolan and Tsuchiya 2005); nutrition (Thunström et al 2015); law (Doremus 2003); Recyling (Manomaivibool and Vassanadumrongdee 2012) (see Noblet et al 2015 for a general discussion). In climate change policies the analyst (implicitly or explicitly) extends present preferences (or willingness to pay (WTP)) into the future by calculating all damages in prices of a base year thus taking existing prices as the basis for aggregation. This ignores changes in relative values of man-made and environmental goods. According to a rather widespread belief, future generations will be richer in material goods but less equipped with environmental amenities. This means that relative prices will shift making environmental amenities more valuable than today. That in turn indicates that the future value of aggregates of market and non-market goods (and GDP is one such aggregate!) will be sensitive to changes in relative prices. Only under very restrictive assumptions – totally unrealistic in modern industrial societies – would such aggregates be invariant to changes in relative prices: the capital to labour ratio in all sectors should be the same meaning that all sectors practically use a uniform technology meaning further that the alleged aggregate of (different) commodities is practically one good (or if you prefer, a Hicksian 'composite good' or a Sraffian 'composite commodity'). We owe these insights to the capital debate of the 1970s often labelled as 'Cambridge versus Cambridge' debate. If we ignore shifts in relative worth of man-made and environmental amenities, climate change damage assessment estimates profoundly underestimate future damages.

A number of authors raise these issues and put the change of relative values / prices and the related question of future preferences in the centre of their analysis (Sterner and Persson 2008). Others allude to it without exploiting it further but an extensive treatment of how to estimate future values is still missing.

## 1.3 Links to the ECONADAPT Work Packages

The present work has benefited from WP1/Task 1b (Review and gap analysis on the costs and benefits of adaptation) on how future values are currently addressed in IAMs. Research presented in this report could be also linked to WP1/Task 1d (Participatory Development of adaptation narratives/scenarios) in order to investigate how existing scenarios address changing preferences and their underlining parameters. It can also be used in the following WPs where estimation of future adaptation benefits is planned

- WP5: Case Study: Disaster Risk Management
- WP6: Case Study: Economic Project Appraisal
- WP7: Case Study: Policy Impact Appraisal
- WP8: Case study: Macro-economic effects of adaptation
- WP9: Case study: International Development Support

Last but not least, the methodology for assessing future trends in values will be integrated into WP10: Toolbox for economic assessment of adaptation

## 1.4 Overview of the report

The report starts by presenting the main challenges in the estimation of climate-related environmental damages in the methodological framework of IAMs (sections 1.1 and 1.2). We proceed to analyse future economic choices with fixed preferences taking into account rising ecosystem scarcity (section 2.1); elasticity of substitution (section 2.2); income elasticities (section 2.3). We then proceed to analyse the role of changing preferences (section 3) before we model future economic choices with fixed and evolving preferences (section 4). We analyse factors affecting future WTP values (section 4.1) and model them in a random walk analytical framework (section 4.2). In section 5 we illustrate the model by estimating the effect of growth rate of WTP values of temperate and boreal forests. We conclude in section 6.

# 2 Challenges in existing estimation of climate change-related environmental damages

## 2.1 IAMs

The Integrated Assessment Models (IAMs) have become a rather widespread tool for the provision of scientific insights in order to confront climate change and to design efficient adaptation policies. To date various classifications of the IAMs have been proposed. Indicatively, Toth (2005) classified the IAMs into two different categories, namely policy evaluation models and policy optimization models. The policy evaluation models exploit simulation techniques in order to assess the effectiveness of policies in the future. To accomplish this, the models quantify policy impacts taking into consideration a variety of modelled variables such as: temperature change, ecosystem changes, sea-level rise etc. On the other hand, the policy optimization models identify the relevant boundaries of policy scenarios as formulated by a set of specific parameters and variables and determine their values through an optimization procedure and the maximization of an objective (welfare) function for alternative paths or policies.

Stanton et al. (2008) classified the IAMs into four different categories, namely: i) welfare optimization models, which maximize the net present value of utility of consumption affected by climate change damages and abatement strategies, ii) general equilibrium models, which model the economy on the basis of sector specific demand and supply functions; general equilibrium models focus on the interactions of energy demand with energy prices, income and climate variables etc., iii) simulation models, which take into account scenarios about the evolution of future emissions and climate conditions and finally iv) cost minimization models, which estimate cost-effective policy scenarios by applying different versions of climate/economy models. Apparently, various overlaps between the sub-groups of the IAMs have been already identified, as many models can be incorporated into more than one category.

Ortiz and Markandya (2009) proposed an additional categorization of the IAMs based on the identified interactions between economic and climate systems by Edwards et al. (2005) (Fig. 1).

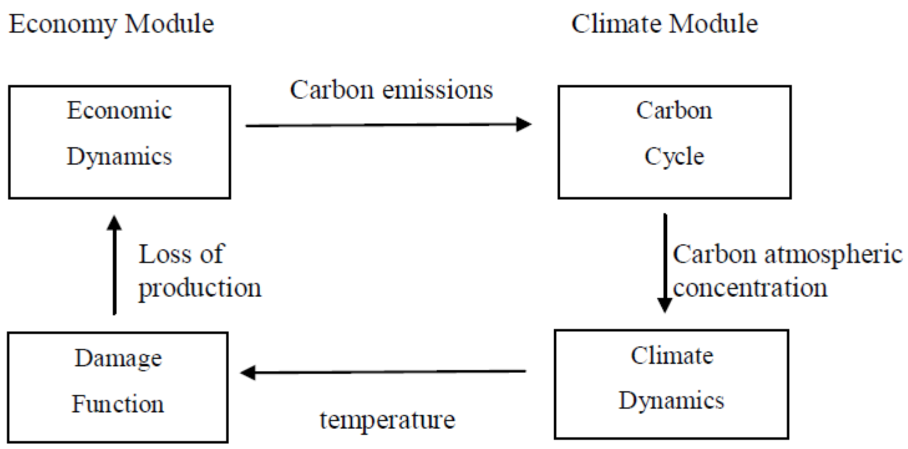



**Fig. 1.** Interactions between economic and climate systems (Source: Ortiz and Markandya, 2009)

Specifically, three different sub-modules can be identified, namely the economic dynamics module, which constitutes the main core of the computable general equilibrium (CGE) models, the energy module, which utilizes a bottom-up approach, and the damage module, which undertakes the analysis of the interactions between climate variations and the triggered impacts in the economy. Taking into consideration this classification, the fully integrated IAMs include all the previous-mentioned modules. The models that do not include an optimization procedure of the economy but implement an analysis with the climate and damage modules were defined as non-CGEs. These models can include also an energy module, and can thus be considered as policy evaluation models according to Toth (2005) or as simulation models according to Stanton et al. (2008). Finally, the CGE models implement an optimization of the overall economy including the energy sector.

The aim of this section is to identify core elements and issues of the IAMs, which could be taken into consideration for the development of the proposed methodological framework. Indicatively, six different models are described briefly in the following section; their selection is based on the degree of utilization and popularity in relevant papers and projects. Nevertheless, although the number of IAMs is relatively high the available information about the damage functions is limited constituting a significant barrier for the conduction of a more detailed and in-depth analysis.

The WITCH model is a hybrid energy-economy model consisting of a climate module and region-specific climate change damage functions, which are connected with the economic system providing the necessary information for the calculations. The WITCH model is a forward-looking model optimizing over a discounted stream of future consumption. The damage function estimates the effects triggered by the increase of the temperature on the economic system. Specifically, a relationship between the atmospheric concentrations of CO2 and the average world temperature has been established, while a quadratic damage function is used in order to estimate the effect on gross output in accordance to the temperature levels. The model is based on the assumption that each region produces one single output that can be used for consumption or investment. The final good is produced using capital, labour and energy services through the implementation of Cobb-Douglas production function and Constant Elasticity of Substitution production function. Climate damage, which is a non-linear function of the difference between current and pre-industrial temperature, calculates the differences between net output and gross output. The optimal consumption is calculated by the intertemporal social welfare function, which is defined as the log utility of per capita consumption, weighted by the regional population. The pure rate of time preference declines from 3% to 2% at the end of the century depicting the fluctuation of historical values of the interest rate. CO2 concentrations have been updated to 385ppm and temperature increase above pre-industrial at 0.76°C in accordance with IPCC 4th Assessment Report (Bosetti et al, 2009).

The equations for the maximization of the utility function, the pure time preference discount factor and the damage function are presented below.

$$W(n) = \sum_t \ U[C(n,t), L(n,t)]R(t) = \sum_t \ L(n,t)\{\log[c(n,t)]\}R(t)$$ (Eq 1)

$$R(t) = \prod_{v=0}^{t}[1+\rho(v)]^{-5}$$ (Eq 2)

$$\Omega(n,t) = 1 + \left(\theta_{1,n}\right) \cdot T(t) + \left(\theta_{2,n}\right) \cdot T(t)^2$$ (Eq 3)

The PAGE2002 model consists of two different damage sectors, namely economic and non-economic. The model is based on the assumption that the impacts can occur only in the context of a temperature increase above a specific tolerable rate of change. Therefore, the regional impact of global warming corresponds to the temperature increase in excess of the adjusted tolerable level; adaptation can increase the tolerable level of temperature increase.

Weights are applied in order to monetize the impacts allowing the comparison and aggregation across economic and non-economic sectors. The weights express the loss of GDP for a benchmark warming of 2.5°C above the tolerable level in each examined sector at the EU level. Impacts are computed for each region, sector, and analysis period as a power function of regional temperature increase above the tolerable level. An adaptive policy can mitigate these triggered impacts.

Equation 4 represents the damage function.

$$WI_{i,d,r=}\left(\frac{I_{i,d,r}}{2.5}\right)^{POW}\cdot W_{d,r}\cdot\left(1-\frac{IMP_{i,d,r}}{100}\right)\cdot GDP_{i,r} \qquad \text{(Eq 4)}$$

Different values can be used in order to discount the costs of both the implemented policy and the climate change impacts. The weighted impacts are aggregated over time with a time-variable discount rate and summed over all regions. The net present value of global warming impacts in the economic and non-economic sectors is calculated according to equation 5:

$$DD=\sum_{i,r}AD_{i,r}\cdot\prod_{k=1}^{i}\left(1+dr_{k,r}\cdot\frac{ric}{100}\right)^{-(Y_k-Y_{k-1})} \qquad \text{(Eq 5)}$$

The same equation can be applied in order to estimate the adaption cost of potential interventions (Hope, 2006).

The MERGE model estimates both market and non-market damages. Typically, it is assumed that a temperature increase of 2.5ºC would double $CO_2$ concentrations over preindustrial levels. Correspondingly, this temperature increase would lead to 0.25% GDP losses in the high-income nations and to 0.50% losses in the low-income nations. It is assumed that at higher or lower temperature levels from this temperature threshold the market losses will be proportional to the change in mean global temperature in relation to the 2000 levels. Concerning non-market damages, the MERGE model is based on a quadratic relation of the expected losses with the temperature increase. The utilized loss functions are based on the determination of two parameters in order to define the willingness-for the avoidance of the temperature increase (catt and hsx). Specifically, the high-income countries might be willing to give up 2% of their GDP in order to avoid a 2.5ºC temperature rise. The determination of hsx is based on the economic loss factor, which represents the fraction of consumption that remains available for conventional uses by households and government. For the case of high-income countries, the loss is quadratic in terms of the temperature increase and as a result the hsx parameter is equal to 1. The loss function can be estimated with equation 6:

$$ELF(x)=\left[1-\left(\frac{x}{catt}\right)^2\right]^{hsx}$$
(Eq 6)

where x is a variable that measures the temperature increase above 2000 levels and catt is a threshold for the temperature indicating that the entire regional product would run out at this level. The corresponding temperature has been estimated equal to 17.7oC. The integration of the loss function into the minimization functions was performed through equation 7:

$$Maximand=\sum_{rg}nwt_{rg}\cdot\sum_{pp}udf_{pp,rg}\cdot log\left(ELF_{rg,pp}\cdot C_{rg,pp}\right) \qquad \text{(Eq 7)}$$

where:

$nwt_{rg}$ denotes the Negisli weight assigned to region rg determined iteratively so that each region will satisfy an intertemporal foreign trade constraint

$udf_{pp,rg}$ is the utility discount factor assigned to region rg in projection period pp

$ELF_{rg,pp}$ is the economic loss factor assigned to region rg in projection period pp

$C_{rg,pp}$ is the conventional measure of consumption (excluding non-market damages) assigned to region rg in projection period pp.

The MERGE model can implement a cost-benefit analysis minimizing the discounted present value of abatement costs and damages (Manne and Richels, 2004).

The CETA-M model utilizes regionalized damage functions introducing benchmark damage from a 2.5°C temperature increase for a doubled $CO_2$ concentration. For the estimation of the climate change impacts, market and non-market damage classes were created. Specifically, the non-market damage class includes the impacts for wetlands loss (even though fisheries loss is included), ecosystem loss, human life, air pollution, migration, natural hazards (even though this is partly a market damage), while the market damage class comprise coastal defence, dryland loss, agriculture, forestry, energy, and water.

For the case of the market damages a relation between GDP and population is used according to equation 8:

$$D_M = a + \beta_1 \cdot GDP \qquad \text{(Eq 8)}$$

where:

a = 3.573223 (SE = 0.97596),

b1 = 0.005327 (SE = 0.000237) and

R2 = 0.992083.

For the case of non-market damages, the estimation is based on the willingness-to-pay to avoid non-market damage [f(y)] and on the population, as presented in equation 9.

$$D_{NM} = f(y) \cdot POP \qquad \text{(Eq 9)}$$

The function f(y) is probably non-linear to income per capita; after a thorough examination though a linear relationship of income per capita to population and GDP (with no constant term) was assumed.

$$\frac{D_{NM}}{POP} = a + \beta_1 \cdot \frac{GDP}{POP} \qquad \text{(Eq 10)}$$

where a = 0.003705, b1 = 0.006017 (SE = 0.000200) and R2 = 0.995566.

As a result, equations 9 and 10 provide functional relationships between income, population, and benchmark damages. Finally, it is necessary to assess the regional temperature changes and to specify the functional relationship between temperature change and resulted damages. To achieve this, it is assumed that warming in the EU and FSU is equal to 1.14 times the global mean temperature rise, while warming in the other regions is 0.86 times the global mean temperature rise. Regarding the relation between regional temperature rise and the benchmark damage it is assumed that it exhibits a quadratic relation (Peck and Teisberg, 1997).

The FUND model performs simplified representations of development, energy use, carbon cycle, and climate. The impacts of climate change are assumed to depend on the impact of the previous year, allowing the assessment of the adaptation to climate change. The FUND model can complement both cost-benefit and cost-effectiveness analyses through the development of separate scenarios, where each scenario is defined initially by exogenous assumptions about various parameters such as (rates of population growth, economic growth, energy efficiency improvements, decarbonisation of energy use etc.). The scenarios of economic and population growth are related to the impacts of climatic change. In this context, all the triggered impacts are monetized. The value of a statistical life is set to be 200 times the annual per capita income, while the monetary values of a loss of one square kilometre of dryland and wetlands are considered as equal on average \$4 million and \$2 million in OECD countries in 1990 correspondingly. Moreover, the FUND model utilizes values about the atmospheric concentrations of carbon dioxide, methane, nitrous oxide and sulphur hexafluoride and calculates the radiative forcing, global mean temperature and sea level rise. The global mean temperature increase in equilibrium by 3.0°C for doubling the CO2 equivalents.

The social cost of any greenhouse gas is computed with equation 11:

$$SC_{r,i} = \sum_{t=2010}^{3000} \frac{D_{t,r}(E_{1950}+\delta_{1950},\dots,E_t+\delta_t) - D_{t,r}(E_{1950},\dots,E_t)}{\prod_{s=2010}^{t} 1+\rho+ng_{s,r}} \Big/ \sum_{t=1950}^{3000} \delta_t \qquad \text{(Eq 11)}$$

$$\delta_t = \{\omega for\ 2010 \leq t < 2020\ and\ 0\ for\ all\ other\ cases\}$$

where:

SCCr is the regional social cost of carbon

r is the region

t and s is the time in years

D are the monetized impacts, E are the emissions

δ are the additional emissions

ρ is the pure rate of time in fraction per year

n is the elasticity of marginal utility with respect to consumption and

g is the growth rate of per capita consumption.

The aggregation of social cost to each region is performed with equation 12:

$$SC_i = \sum_{r=1}^{16} \frac{Y_{2010,ref}}{Y_{2010,r}}^{\varepsilon} SC_{r,i} \qquad \text{(Eq 12)}$$

where

$SC_i$ is the global social cost of greenhouse gas i

$Y_{2010,ref}$ is the average per capita consumption in the reference region in 2010

$Y_{2010,r}$ is the regional average per capita consumption in 2010 and

ε is the rate of inequity aversion (ε=0 in the case without equity weighting and ε=n in the case with equity weighting).

Finally, the unitless damage potential of greenhouse gas - corresponding to the relative marginal damage of greenhouse gas with respect to the social cost of carbon dioxide - is estimated with equation 13 (Waldhoff et al. 2014):

$$DP_i = \frac{SC_i}{SC_{CO_2}} \quad \text{(Eq 13)}$$

The DICE model is a simplified analytical and empirical model facilitating the analysis of various economic, policy, and scientific aspects of climate change. The DICE/RICE models are primarily designed as policy optimization models; the option to use them as simple projection models is though available. The main target is the maximization of the economic objective function, which refers to the economic utility associated with a path of consumption and its relation with the population. The relative importance of the different generations is affected by the pure rate of social time and the elasticity of the marginal utility of consumption.

The DICE model assumes that economic and climate policies should be designed to optimize the flow of consumption over time integrating both market and non-market values. The target of the DICE model is to maximize the social welfare function (W), which is the discounted sum of the population-weighted utility of per capita consumption. According to equation 14, the calculation of social welfare is based on the per capita consumption (c(t)), the population and the labour inputs (L(t)) and the discount factor (R(t)).

$$W = \sum_{t=1}^{T_{max}} U[c(t), L(t)]\, R(t) \quad \text{(Eq 14)}$$

The pure rate of social time preference (ρ) is the discount rate that provides the welfare weights on the utilities of different generations. The required outputs are measured in purchasing power parity (PPP) exchange rates using the IMF estimates, while their estimation is performed at regional level and then aggregated at global level. The regional and global production functions are assumed to be constant-returns-to-scale Cobb-Douglas production functions in capital, labour and Hicks-neutral technological change. The estimation of the global output is performed with equation 15:

$$Q(t) = [1 - \Lambda(t)]\, A(t)K(t)^{\gamma}L(t)^{1-\gamma}/[1 + \Omega(t)] \quad \text{(Eq 15)}$$

where:

Q(t) is the net output of damages and abatement

A(t) is the total factor productivity (of the Hicks-neutral variety) and

K(t) the is capital stock and services.

The additional variables in the production function are Ω(t) and Λ(t), which represent climate damages and abatement costs respectively. The damage function is shown in equation 16:

$$\Omega(t) = \psi_1 T_{AT}(t) + \psi_1 [T_{AT}(t)]^2 \qquad \text{(Eq 16)}$$

It should be noted that the damage function has been greatly simplified in comparison with earlier versions of the DICE/RICE model because some of the employed elements were considered as out-dated and unreliable.

The last version of the DICE model uses estimates of monetized damages from the Tol (2009) survey as the starting point. Due to the fact that some factors are uncertain and difficult to be modelled (such as the economic value of losses from biodiversity, ocean acidification, and political reactions, extreme events etc.) an adjustment of 25% of the monetized damages is used in order to incorporate the non-monetized impacts. Moreover, the current version assumes that damages are a quadratic function of temperature change and no sharp thresholds or tipping points are taken into consideration.

The abatement cost function - shown in equation 17 - is a reduced-form type model where the costs for emissions reductions are considered a function of the emissions reduction rate μ(t).

$$\Lambda(t) = \theta_1(t)\mu(t)^{\theta_2} \qquad \text{(Eq 17)}$$

According to equation 17, the abatement costs are proportional to the output and to a power function of the reduction rate (Nordhaus and Sztorc 2013).

Summarizing, the analysis of the six different IAMs revealed similarities and differences for the most crucial elements of these models. The conclusion from Tol and Frankhauser (1998) confirms that the majority of IAMs calculate the climate change damages in a reduced or simple form associating the triggered damages with the average global surface air temperature while the exclusion of the non-market or intangible damages from the analysis leads to completely different outcomes. Moreover, the critical issue of the future preferences is also ignored during the performed analysis for the calculation of non-market damages.

## 2.2 Current IAMs pitfalls

Based on the analysis presented in Section 2.1, it is acknowledged that IAMs are currently faced with a number of challenges regarding the estimation of climate change-related damages to environmental goods and services. To put it in a nutshell:

- A perfect substitutability between man-made and ecosystem goods and services is commonly assumed.
- Climate change environmental damages are based on market prices (i.e. they are estimated on a GDP basis through consumption reduction).
- Future changes in relative prices/values of man-made goods and ecosystem services are ignored.

The above points of criticism towards IAMs are already noticed in the relevant literature regarding these issues (e.g. Horowitz, 2002; Ayong Le Kama and Schubert, 2004; Hoel and Sterner, 2007; Sterner and Persson, 2008; Jacobsen et al., 2013). Some of the arguments are not new; for example, Fisher and Krutilla (1975) proposed a method for evaluating resource development projects in natural environments in which the net result of a change in the discount rate used depends upon empirical magnitudes in the economy, including the response of investment to the change, the elasticity of demand for natural resource inputs to productive investment and so on. The authors also noticed that there are perceived differences between alternative natural environments for the production of recreation and other preservation-related services.

### 2.2.1 Assumption of perfect substitution between environmental and market goods

As mentioned by Sterner and Persson (2008), the assumption of perfect substitutability - that is the hypothesis that climate change impacts can be balanced on a one-to-one basis by an increased consumption of material goods - is implicit in all IAMs used in the analysis of climate change policy and the estimation of the social cost of carbon, e.g. DICE, PAGE, FUND (e.g. Tol, 1999; Hope, 2006; Nordhaus and Boyer, 2000; Nordhaus and Sztorc, 2013). The implicit rationale of this assumption is that despite climate damages and the loss of ecosystem services we may enjoy a higher level of welfare in the future as long as we are compensated by an equal amount of welfare gain from material consumption. Nevertheless, this assumption jeopardizes future generations' ability to meet their needs, in at least two different ways, as discussed hereinafter.

First, as Sterner and Persson (2008) argue, if there are limits to the substitutability between material consumption and environmental services, climate change-related analyses need to consider the content of future growth; if growth is unbalanced, we could experience an increase in the relative prices of those goods or services which would become scarcer. To illustrate their point, the authors amended the DICE model by changing the equation of utility function to include an extra parameter that determines how consumption of environmental good changes over time in response to climatic change. In this way, utility is dependent not only on the consumption of material goods but also on environmental goods. Further, they assumed that: (a) today people allocate 10% of total expenditures to the consumption of environmental goods and services; (b) the substitutability between market and nonmarket goods, which is expressed by the elasticity of substitution, is 0.5, i.e. if the relative price of the environmental good increases by 1%, then the purchase of environmental goods will decline by 0.5% relative to the purchase of other goods; and (c) nonmarket impacts are equal, in economic terms, to the impacts on material consumption. These results showed that accounting for relative price changes can have a detrimental effect on necessary abatement that is on the same order of magnitude as changing the discount rate; that entails stronger support for firms and immediate abatement measures.

Second, the assumption of perfect substitution may raise serious concerns from theoretical, practical and ethical perspectives, as it is linked to the definition of sustainable development, i.e. "…the development that meets the needs of the present without compromising the ability of future generations to meet their own needs…" (WCED, 1987). As Traeger (2007) mentions, the substitutability between environmental services and material consumption is the most important distinction between the concepts of "weak" and "strong" sustainability. The notion of "weak" sustainability allows for a substitution between environmental and man-made capital. On the other hand, "strong" sustainability requires a non-declining value or physical amount of natural capital and its service flows. The former is mainly concerned with the preservation of a non-decreasing overall welfare. The latter requires a non-declining value or physical amount of natural capital and its service flows (Traeger, 2007). Defenders of "strong" sustainability argue that substitution possibilities between man-made goods and natural goods and services are either limited or ethically indefensible, and, for this reason, we need to sustain a certain stock of "natural capital". To this end, it is also argued that under the concept of "weak" sustainability man-made and natural capital are basically substitutes, whereas under the notion of "strong" sustainability man-made and natural capital are basically complements (Daly, 1995). From this standpoint, Ayres at al. (1998) mention that "…the recognition that natural resources are essential inputs in economic production, consumption or welfare that cannot be substituted for by physical or human capital, or the acknowledgement of environmental integrity rights in nature…" The 'environmental integrity rights' are referred to as intrinsic values of nature, that is values, which are generally regarded to be non-anthropocentric concepts, based on moral, ethical or religious considerations (Traeger, 2007). This issue is further discussed in the next section.

Apart though from ethical and theoretical arguments, there are also practical considerations associated with the assumption of perfect substitution. Traeger (2007) analyses how limited substitutability in consumption between environmental and man-made goods affects the social discount rates. This issue receives much attention in the literature under the concept of declining discount rates on the basis of intergenerational justice or behavioural aspects (e.g. Newell and Pizer, 2003; Groom et al., 2005; Pezzey, 2006; Hoel and Sterner, 2007; Grijalva et al., 2013). The usual intuition is that expressed by Groom et al. (2005): "….using a declining discount rate would make an important contribution towards meeting the goal of sustainable development…" Nevertheless, the debate is still open. For instance, Traeger (2007) argues that the notion of strong sustainability presupposes that optimal social discount rates should be growing over time.

### 2.2.2 Pricing of nonmarket goods

Even under the assumption of perfect substitutability between market and environmental goods, current practices implemented in IAMs are far from being consistent with the concept of environmental valuation. The main reason for that, as discussed hereinafter, is that market prices don't reflect the true value (i.e. social worth) of environmental goods and services because of "market failures" (e.g. Turner et al., 1994; Freeman III, 2003).

From an economic point of view, the value of nonmarket assets, like the environmental ones, reflects the changes to economic welfare from small or marginal changes in the quality or the availability of the asset (e.g. Turner et al., 1994). This monetary measure is based on the concept of Total Economic Value (TEV). In instrumentally valuing a nonmarket resource, the TEV can be usefully disaggregated into use values and non-use values.

Use values involve direct use (i.e. actual use of an environmental good or service for commercial purposes or recreation); indirect use (i.e. benefits from ecosystem services and functions rather than directly using them); and option values (i.e. the value of ensuring the option to use a resource in the future, which could be seen as an insurance premium) (Freeman III, 2003; TEEB, 2010; Brouwer et al., 2013). Non-use values derive from the knowledge that the environment is maintained and include altruistic values, which are related to the use of environmental goods and services from others; bequest values that reflect values that people may hold for ensuring that their heirs will be able to use a natural resource in the future; and existence values which reflect the fact that people value resources for moral reasons, unrelated to current or future use (DEFRA, 2007). Non-use value is closely linked to ethical concerns, as mentioned above, although for some analysts it stems ultimately from self-interest (Kontogianni et al., 2012).

Several environmental valuation techniques exist, which differ in data requirements, assumptions regarding economic agents, and values that they are able to capture. Broadly speaking, valuation techniques are divided into the following three categories (TEEB, 2010):

- Direct market valuation approaches (market price-based, cost-based, and production functions), e.g. replacement cost, damage avoided cost, substitute (or alternative) cost, and productivity change cost;
- Revealed preference approaches, i.e. the Travel Cost Method (TCM) and the Hedonic Pricing Method (HPM), which elicit preferences from the actual behaviour of individuals based on market information, and;
- Stated preferences approaches that attempt to elicit individuals' preferences directly by means of social surveys on hypothetical changes in the quantity or quality of environmental and/or social goods and services. The main types of stated preference techniques are: the Contingent Valuation method (CVM) and the Choice Modelling (CM). Furthermore, Group Valuation (GV) approaches are also considered in this category. The latter combine stated preference techniques with elements of deliberative processes from political science.

The selection of appropriate valuation technique is mainly determined by the type of good or service being valued, since each approach has its own advantages and disadvantages. Direct market valuation approaches rely on data, which are easier to obtain. Nevertheless, if markets do not exist for the goods and services under question, then these approaches are not available (TEEB, 2010). Furthermore, and more importantly, the direct market does not reflect the total value of the good due to the difference between the market price and people's willingness to pay, which is known as Consumer Surplus (CS). Finally, direct market valuation approaches, like revealed preference approaches, are not capable of capturing non-use values (e.g. Freeman III, 2003; Brouwer et al., 2013). For all these reasons, it is argued that nonmarket damages due to climate change are underestimated when calculations are based solely on changes in the output of the economy.

### 2.2.3 Nonmarket values and preferences

Nonmarket value refers to small changes in the state of the environment, and not the state of the environment itself (TEEB, 2010). The estimates are based on people willingness-to-pay (WTP) – the maximum amount of money in order to avoid an environmental degradation and its consequences on health, amenity, etc. - or their willingness-to-accept (WTA) – the minimum compensation in order to endure the environmental impacts incurred (Turner et al., 1994; Freeman III, 2003). In this regard, the value of environmental assets is individual-based and subjective as well as context- and state-dependent (Goulder and Kennedy, 1997; Nunes and van den Bergh, 2001). To this end, the estimates of nonmarket values reflect only the current choice pattern and are affected by socio-economic-ecological conditions such as the state of the environment, the available income, attitudes, opinions and beliefs, and expectations about the future (Barbier et al., 2009). It is evident that a change in socio-economic-ecological conditions might severely affect the estimated values (TEEB, 2010).

The importance of incorporating preferences concerning environmental service flows and environmental quality into the sustainability analysis is pointed out by Pearce et al. (1997). These concerns are not new in the literature but they gain a new urgency, especially when the nonmarket impacts of climate change are put at the centre of the discussion. For example, Witsenhausen (1974) pointed out the problem of preference evolution mentioning that: "…Taking the initial preference relation as absolute and permanent can lead to commitments which will be harshly judged in a later climate… This difficulty has long been perceived but in careful treatments of dynamic utility theory it is only mentioned to be assumed away…". Changing values with respect to the environment were the focus of early work on discount rates for environmental projects by Fisher and Krutilla (1975). They suggested that evolving preferences could be captured by assuming that the "present" WTP (WTP0) for the environment would change at some pre-determined rate, say α:

$$WTP_t = WTP_0 e^{\alpha t} \qquad \text{(Eq 18)}$$

Where:

$WTP_0$ is the present WTP for an environmental good or service

$WTP_t$ is the future WTP for the environmental good or service at time t

α is the growth rate of the WTP value for the environmental good or service

t is the time in years (assuming that α and discount rates are expressed on a yearly basis)

The present value (PV) of $WTP_t$, assuming a social discount rate s and continuous compounding, is estimated as follows:

$$PV_{WTPt} = WTP_0 e^{\alpha t} / e^{st} \qquad \text{or}$$
$$PV_{WTPt} = WTP_0 / e^{(s-\alpha)t} \qquad \text{(Eq 19)}$$

Fisher and Krutilla (1975) define (s-α) as the 'environmental' discount rate, suggesting that the change in the value of the environmental goods can be captured by this 'net' discount rate. According to Fisher and Krutilla (1975) and Horowitz (2002) two factors are likely to determine the growth rate of WTP (i.e. α): income growth and changes in environmental quality, that is, the effect of resource scarcity. To put it simply, assuming that the environment in broad sense is a normal good, future WTP values will be higher than present ones if future incomes are higher. The increase in WTP owing to the growth of income depends on the income elasticity of the good (Gravelle and Smith 2000; Horowitz, 2002; Groom et al., 2005). Similarly, the scarcity of environmental goods and services will likely increase future WTP for environmental goods assuming a negative price elasticity of demand. Price and income elasticities are commonly used to examine consumers' behaviour towards certain goods and services and to study whether these goods and services are price inelastic or elastic, normal or luxury, etc. Nevertheless, when it comes to the environment neither price nor income elasticity of demand are straightforward to estimate, since welfare estimates are usually defined from the indirect utility function. Hence, due to methodological restrictions in the environmental 'market', the demand function of the good or service and, consequently, the price and income elasticities of demand cannot be calculated directly (Hökby and Söderqvist, 2003). Thus, in many cases the income elasticity of WTP is estimated instead, using the WTP function when income is included as explanatory variable. The latter is the appropriate concept for investigating the distributional impacts of examined policies (Flores and Carson, 1987).

Besides the above-mentioned factors, i.e. income growth and environmental scarcity, WTP values are influenced by the substitution elasticity between natural and man-made capitals. As mentioned by Sterner and Persson (2008), when nonmarket goods are perfectly substitutable with market goods nonmarket damages can be included in consumption directly, since elasticity of substitution is 1 or more, and the effects of changing WTP values of environmental goods are weakened substantially. However, when there are limits to the substitutability between market and environmental goods (i.e. when elasticity of substitution is less than 1), WTP would be expected to rise with increasing scarcity. Similar results are reported by Hoel and Sterner (2007) and Krysiak and Krysiak (2006). The elasticity of substitution varies considerably from one environmental good or service to another, as well as between individuals, therefore it is hard to provide a good empirical estimate for the elasticity of substitution and particularly hard to say how it would evolve over time (Sterner and Persson, 2008).

Finally, future WTP values will be affected by the attitude of future generations towards environmental assets. Their attitude may well be different from ours for many reasons, some of which are obvious and others unpredictable, since the formation of preferences involves complex and interlinked economic, social and moral determinants (Ayong Le Kama and Schubert, 2004). There are good reasons to suspect that people will care more about the environment in the future (greening of preferences), leading to higher WTP values. Yet, this is not definite as growing materialism may result in lower WTP values owing to less concern about the effects of environmental damage on other people, ecosystem, and future generations.

All in all, ignoring the effect of changing WTP values for natural capital could seriously undermine the validity of damage assessments relating to climate change impacts. As Sterner and Persson (2008) conclude after modifying DICE model to account for environmental scarcity:

"…Even if the climate damages in the DICE model used in the numerical exercise above are doubled to account for a wider range of nonmarket impacts, following the results in the Stern Review, we would argue that these impacts are still comparatively low. As discussed above, total damages in our modified DICE model amount to just over 2 percent of GDP, for a temperature increase of 2.5◦C. As noted by Manne et al. (1995), US expenditures (which should be smaller than averted damages) on environmental protection totalled about 2 percent of GDP in 1995. Thus, the suggestion of current IAMs that we should be willing to spend much less on climate protection, one of the biggest environmental problems facing humanity, seems implausible….We believe that it is exactly the nonmarket effects of climate change that are the most worrisome. If we focus on the risk for catastrophes, as Weitzman suggests, then we believe the main effect of climate change will not be to stop growth in conventional manufacturing, but rather to damage our ability to enjoy some vital ecosystem services..."

Nevertheless, the issue of non-constant WTP values and the effect that they could have on the 'environmental' discount rate - as defined by Fisher and Krutilla (1975) - is acknowledged and then left aside (Sterner and Persson, 2008 providing examples from Arrow et al., 1996; Nordhaus 1997; Lebègue et al. 2005; and Gollier 2007). Therefore, current IAMs create the feeling that the economic damages due to the effects of climate change are being trivialized (Parry et al., 2001).

Bearing in mind the above remarks, our analysis focuses on developing a stochastic model for estimating the growth rate of WTP values (α). To this end, the following sections discuss in detail the factors affecting α; present the simulation functions constructed using random walk-based stochastic models for estimating α; and provide an illustrative example including sensitivity analysis and probabilistic simulations.

# 3 Future economic choices with fixed preferences

## 3.1 Rising ecosystem scarcity

### 3.1.1 Price elasticity of demand

As mentioned in the previous sections, the estimation of price elasticities in environmental goods and services is not always as simple as in the case of market goods. The literature provides several studies that attempt to estimate price elasticities of environmental assets. Not surprisingly, the majority of them deal with environmental resources that can be easily deemed as market goods, e.g. residential or irrigation water.

Thomas and Syme (1988) conducted a Contingent Valuation (CV) survey to estimate price elasticity of demand for public water supply. According to their econometric results price elasticities varied from -0.1 to -0.58. Nevertheless, the higher range of elasticities was obtained from a model with low R and positive serial correlation amongst residuals. In the three best models the range of price elasticities was between -0.1 and -0.43.

Hewitt and Hanemann (1995) used a discrete/continuous choice model of the residential water demand under block rate pricing and examined a dataset from a previous published study. Their study concluded that the discrete/continuous choice model results in much more elastic estimates, since the price elasticity of demand falls in the range of -1.53 to -1.629.

Esprey et al. (1997) performed a meta-analysis to explain the variation in the price elasticity of residential water demand across different studies. The meta-analysis was based on a review of 24 journal articles published between 1967 and 1993. Their findings suggest that price elasticity estimates range from -0.02 to -3.33. The average price elasticity is -0.51 whilst about 90% of the estimates are between 0 and -0.75. They conclude that the most important influences on the price elasticity of demand for residential water seem to be evapotranspiration rates, rainfall, the pricing structure and the season. Furthermore, it seems that there are differences between short-run and long-run price responsiveness and between residential and commercial demand.

Pint (1999), using fixed effects and maximum likelihood techniques, estimated residential water demand during the California drought. The study estimated that price elasticity of demand ranges between -0.14 and -1.24.

Dalhuisan et al. (2003) conducted a meta-analysis of variations in price and income elasticities of residential water demand. As they note, the price elasticity estimates in the literature vary between –7.5 and +7.9. The distribution of price elasticities has a sample mean of –0.43, a median of –0.35, and a standard deviation of 0.92. Among the studies examined, approximately 90% of the price elasticity estimates ranged between 0 and -0.75.

Arbués et al. (2003) examined differences in the specification of water demand models, and analysed several tariff types and their objectives through an extensive literature review. Among the issues addressed is that of price elasticity of demand. The authors find that the magnitude of price elasticity estimates varies both with the econometric techniques applied and the type of data used. Table 1 presents the price elasticities from selected studies after 1990 for different price specifications.

Scheierling et al. (2006) conducted a meta-analysis to investigate sources of variation in empirical estimates of the price elasticity of irrigation water demand. Elasticity estimates are drawn from 24 studies reported in the U.S. from 1963 to 2004, including mathematical programming, field experiments, and econometric studies. A total of 73 price elasticity estimates were obtained. The estimated mean price elasticity is 0.48 and the median 0.16 (both in absolute terms), implying that irrigation water demand is generally price inelastic. The standard deviation is relatively large (0.53), with estimates ranging from 0.001 to 1.97 (in absolute terms).

Worthington and Hoffman (2008) also provide a synoptic survey of empirical residential water demand analyses conducted in the last 25 years. To this end, they analyse both model specification and estimation, and discuss the outcomes of the analyses. Price elasticity estimates are generally found in the range of 0 to 0.5 in the short run and 0.5 to 1 in the long run (in absolute terms). They also estimate income elasticities, which are of a much smaller magnitude (0.01 to around 0.8 and positive, with one exception by Agthe and Billings (1980) who estimated short-run income elasticities between 1.33 and 2.07, and long-run income elasticities in the range of 1.97–2.77.

Yoo et al. (2014) use a regression model including direct measures of changes in water prices to distinguish between the short- and long-run price elasticity of residential water demand in Phoenix, Arizona. Their estimate of the "short-run" price elasticity over the interval 2000-2002 is -0.661 and that of the "long run", over the interval 2000-2008, is -1.155. These estimates are consistent with previous findings; however, they are higher than many others reported in the literature.

Borcherding and Deacon (1972), using a logarithmic model of per capita expenditure, conducted a collective study to estimate the parameters that govern the demand of public services in the U.S. for various public goods, including parks and recreation. The regression results show price elasticities of demand to be less than one in absolute terms.

Bergstrom and Goodman (1973) developed a method for estimating demand functions of individuals for municipal public services by regressing the expenditures of municipalities on specific services, among them parks and recreation. According to the results, the price elasticity of demand is higher than 1 (around 1.13) on absolute terms given that tax share elasticity is -0.19.

**Table 1.** Price elasticities for different price specifications

| **Price specification** | **Study** | **Price elasticity** |
|---|---|---|
| Nordin specification (marginal price and difference) | Hewitt and Hanemann (1995) | −1.57 to −1.63 |
| | Barkatullah (1996) | −0.23 to −0.28 |
| | Agthe and Billings (1997) | −0.39 to −0.57 |
| | Dandy et al. (1997) | −0.12 to −0.86 |
| | Corral et al. (1998) | −0.11 to −0.17 |
| | Renwick and Archibald (1998) | −0.33 to −0.53 |
| | Renwick and Green (2000) | −0.16 |
| | Martínez-Espiñeira (2002b) | −0.12 to −0.28 |
| Marginal price | Schneider and Whitlatch (1991) | −0.11 to −0.262 |
| | Lyman (1992) | −0.39 to −3.33 |
| | Martin and Wilder (1992) | −0.32 to −0.60 |
| | Nieswiadomy (1992) | −0.02 to −0.17 |
| | Nieswiadomy and Cobb (1993) | −0.17 to −0.29 |
| | Hansen (1996) | −0.003 to −0.1 |
| | Kulshreshtha (1996) | −0.23 to −0.78 |
| | Höglund (1999) | −0.10 |
| | Pint (1999) | −0.04 to −1.24 |
| Average price | Griffin and Chang (1990) | −0.16 to −0.38 |
| | Rizaiza (1991) | −0.78 to +0.18 |
| | Martin and Wilder (1992) | −0.49 to −0.70 |
| | Nieswiadomy (1992) | −0.22 to −0.60 |
| | Stevens et al. (1992) | −0.10 to −0.69 |
| | Nieswiadomy and Cobb (1993) | −0.45 to −0.64 |
| | Point (1993) | −0.167 |
| | Kulshreshtha (1996) | −0.34 to −0.96 |
| | Höglund (1999) | −0.20 |

| | | |
|---|---|---|
| | Nauges and Thomas (2000) | −0.22 |
| Other specifications | Griffin and Chang (1990) | −0.01 to +0.035 |
| | Nieswiadomy (1992) | −0.29 to −0.45 |
| | Renzetti (1992) | −0.01 to −0.65 |
| | Nieswiadomy and Cobb (1993) | −0.319 to −0.637 |
| | Bachrach and Vaughan (1994) | −0.03 to −0.47 |
| | Arbués et al. (2000) | −0.002 to −0.655 |

Source: Arbués et al. (2003), modified by the authors

Rollins and Lyke (1998) conducted a CV study to address the debate over sensitivity of existence values to scope tests. The existence value of remote wilderness parks was examined as case study. Their findings illustrate that a ‘WTP versus number of parks’ curve is clearly increasing and concave as the theory suggests. In addition, they estimated marginal WTP from the total WTP values where possible. The marginal value of the first park is equivalent to the value for one park, estimated at CAN$105.45. The marginal WTP for a second park CAN$56.40 and the marginal WTP for two more parks (the third and fourth) is equal to CAN$29.72, so the average WTP for each of these parks is CAN$14.86. From the phone survey, it was estimated that adding six more parks after four is CAN$26.33, that is an average of CAN$5.26 for each of these six parks (Fig. 2).

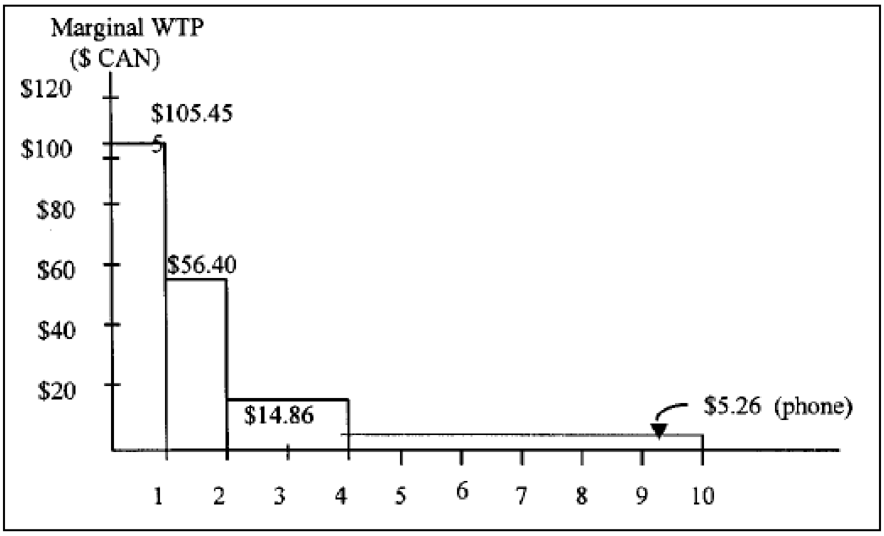



**Fig. 2.** Diminishing marginal WTP per household (Source: Rollins and Lyke, 1998)

Using the standard formula of price elasticity of demand εp, the following WTP elasticities estimates are derived:

- From the first to the second park: εWTP = ΔQ%/ΔP% = [(2-1)/1]/(56.40-105.45)/105.45]= -2.15
- From the second to the fourth park: εWTP= ΔQ%/ΔP% = [(4-2)/2]/(14.86-56.40)/56.40]= -1.36
- From the fourth to the tenth park: εWTP= ΔQ%/ΔP% = [(10-4)/4]/(5.26-14.86)/14.86]= -2.32

Rosenberger and Stanley (2010) conducted a meta-regression analysis of own-price elasticity of recreation demand estimates in the U.S. Their survey was based on a research database currently containing 329 documents from 1958 to 2006 based on data collected from 1956 to 2004 that jointly provide 2,705 estimates of recreation use values. Own-price elasticity estimates are only derived from 119 travel cost studies documented from 1960 to 2006 studies, providing 610 estimates. The raw average elasticity is unitary elasticity (-0.997), while the median elasticity is inelastic (-0.567). However, the precision effect estimate with standard errors shows the standard error-corrected empirical elasticity is -0.158; that is, recreation demand is not very inelastic.

Hökby and Söderqvist (2003) using data from five Swedish CV studies provide estimates of income and price elasticities of demand for reduced marine eutrophication effects in the case of the Baltic Sea. Point estimates indicate that reduced marine eutrophication effects can be classified as a necessity and an ordinary and price-elastic service. More specifically, the point estimate of price elasticity of demand is –1.86 (95% confidence interval: –2.08, –1.64). According to the confidence interval for the base case, a 1% increase (decrease) in price would result in about a 1.6–2.1% decrease (increase) in the demand for reduced eutrophication effects.

Khan (2009) used data from two studies and implemented survey-based data approach, similar to that of Hökby and Söderqvist (2003), for modelling the demand for improved environmental quality of two National Parks in Northern Pakistan and for estimating income and price elasticities. The estimates indicate that improved environmental quality effects can be described as a luxury and an ordinary and price elastic service. As regards price elasticity, the point estimate is -2.15 (95% confidence interval: -2.45, -1.85). According to the confidence interval for the base case, a 1% increase (decrease) in price would result in about a 1.8–2.4% decrease (increase) in the demand for quality improvements.

Khan (2012) used a CV survey to ask a sample of respondents about their awareness and WTP for safe drinking water in Northern Pakistan Using the same approach, as in previous study, reports identical estimates for quality effects of drinking water. More specifically, the study provides a 95% confidence interval for price elasticity around the point estimate of -2.15 (-2.45, -1.85). According to the confidence interval, a 1% change in the price of water would result in more than 1% change (in the opposite direction) in the demand for improvement in the quality of drinking water.

### 3.1.2 Environmental scarcity and Hotelling's rule

When a natural resource is depleted faster than it can be replenished, it can be considered as non-renewable. In such cases, an alternative way to estimate the effect of scarcity over future WTP values is to make use of the "Hotelling's rule" which suggests that in an optimal, intertemporal consumption path the shadow price of the stock of an exhaustible resource should grow at the rate of interest (Hotelling, 1931). Hotelling's model provides five major insights (Cairns, 2014):

- Exhaustible resources are a form of capital.
- The price of the resource is determined in a dynamic equilibrium that regulates both the flow of the resource to market and the holding of resources as assets.
- The timing of decisions is of central significance and warrants careful analysis.

- Resources are subject to the usual market failures, viz. monopoly and externality.
- Exhaustibility in itself does not entail a special form of market failure. In particular, competitive markets are not subject to a myopic inability to allocate an exhaustible resource in way that efficiently balances the interests of the present and the future.

Hotelling's theory is fundamental in two aspects: first, it defines the optimal rate of extraction of exhaustible resources, and second, it introduces the concept of scarcity rent. Hotelling's rule presumes the validity of the following conditions for the examined period (Minnitt, 2007):

- The resource owner's objective is to maximize the present value of its current and future profits.
- This extraction takes place along an efficient path in competitive market equilibrium, i.e. the owner of the resource has no control over the price he receives for his production.
- The production is not constrained by existing capacity; the owner may produce as much or as little as he likes at any time during the life of the resource.
- The resource has a capitalized value.
- The resource stock is homogenous and there is no uncertainty about the size and the quality of the resource. In addition, current and future prices and extraction costs are known,
- The costs of extraction do not change as the resource is depleted.

Since the quantity of the resource is limited, the resource consumed today will be not available for future generations. Therefore, a future value is lost due to extracting the resource today, which is known as the opportunity cost. According to Hotelling, the latter equals the discounted present value of the future profit, which will be lost due to extraction of the resource in the present. Hence, the owner will be indifferent between extracting an additional unit of the resource today at price P0, or in the future at price Pt as long as the following condition, known as 'Hotelling r-per cent rule', holds (Minnitt, 2007):

$P_t = P_0 e^{rt}$ (Eq 20)

where Pt is the price of the resource at the given time, P0 is the price at present time, t is the time elapsed and r is the market discount rate that is used for the calculation of the present value.

Therefore, the natural resource price should be increasing over time if marginal costs are constant, and the rate of price increase (dP/P) equals with the market discount rate:

$dP/P = r$ (Eq 21)

However, as Lin and Wagner (2007) remark - based on the actual behaviour of mineral prices over time - the Hotelling's model assumes that the costs of extraction do not depend on the stock of reserve remaining in the ground. It is plausible though that extraction costs increases the more resource units are extracted and fewer reserves remain. As a result of the dependence of extraction cost on the stock of remaining reserve, the shadow price rises less than the rate of interest, but the market price still increases over time (Tietenberg, 1996). Furthermore, technological progress may occur causing the extraction cost to decrease over time.

In the case of environmental and goods and services, the value (price) of the resource is expressed via society's WTP to preserve the resource and the market discount rate, r, can be replaced by the social discount rate, s. Hence, according to Hotelling's rule, WTP must grow at the social discount rate, s, as follows:

$$WTP_t = WTP_0 \, e^{st} \qquad \text{(Eq 22)}$$

where:

$WTP_0$ is the present WTP for an environmental good or service

$WTP_t$ is the future WTP for the environmental good or service at time t

s is the social discount rate

t is the time in years

Consequently, the present value of WTPt, assuming a social discount rate s, is estimated as follows:

$$PV_{WTPt} = WTP_0 e^{(s-s)t} = WTP_0 \qquad \text{(Eq 23)}$$

### 3.1.3 Elasticity of substitution

Consider a utility function of the following simple form (Hoel and Sterner, 2007; Sterner and Persson, 2008):

$$U(C) = \frac{1}{1-\eta} C^{1-\eta}, \text{ for } \eta > 0 \text{ and } \eta \neq 1 \qquad \text{(Eq 24)}$$

where:

U(C) is a concave, twice differentiable utility function

C is the consumption

η is the elasticity of marginal utility of consumption

If the elasticity of marginal utility of consumption (η) is constant, then the appropriate social discount rate, s, can be expressed by the following well-known relation, which is often called the "Ramsey" rate:

$$s = \delta + \eta g \qquad \text{(Eq 25)}$$

where:

δ is the rate of pure time preference

η is the elasticity of marginal utility of consumption

g is the rate of growth of consumption per capita

To put the environment into perspective, suppose that E represents some aggregate measure of the environmental quality and C is the aggregate measure of all other goods. Hence, the utility function takes now the form U=U(C, E). To illustrate the role of the elasticity of substitution between market and non-market goods we have to consider a constant elasticity of substitution (CES) utility function of the following form (Hoel and Sterner, 2007):

$$U(C,E) = \frac{1}{1-\eta}\left[(1-\gamma)C^{1-\frac{1}{\sigma}} + \gamma E^{1-\frac{1}{\sigma}}\right]^{\frac{(1-\eta)\sigma}{\sigma-1}} \qquad \text{(Eq 26)}$$

where:

U(C,E) is a concave, twice differentiable utility function

C is the consumption

E is the environmental quality

η is the elasticity of marginal utility of consumption

γ is a measure of the share of expenditures that consumers would use on environmental quality if environmental quality was a good that could be purchased, i.e. it determines the share consumption of nonmarket goods in the utility function

σ is the elasticity of substitution, which is positive

This elasticity of substitution σ corresponds to the decline by σ% in the purchase of environmental goods if their price increases by 1% relative to the price of other consumer goods. The lower is the elasticity of substitution the less willing are consumers to substitute away from environmental quality as its price increases.

The valuation of the environmental good E is given by UE/UC and expresses how much consumption must increase to just offset a unit of deterioration in environmental quality (Hoel and Sterner, 2007). In this case, the relative change in the “price” dp of environmental quality is given by:

$$\mathrm{d}p = \frac{\frac{d}{dt}(\frac{U_E}{U_C})}{(\frac{U_E}{U_C})} = \frac{1}{\sigma}(g_c - g_E) \qquad \text{(Eq 27)}$$

where:

$g_C$ is the relative growth rate of consumption

$g_E$ is the relative growth rate of environmental quality

According to the above formula, the price change depends on the growth rate of consumption, the growth rate of environmental quality and the elasticity of substitution. If consumption increases over time while environmental quality is constant or declines, the price change, dp, will be positive and will be larger the smaller is the elasticity of substitution (Hoel and Sterner, 2007). In addition, as Krysiak and Krysiak (2006) note, for constant nature and capital stocks, an increase in the elasticity of substitution increases the absolute change in the maximal stock of nature at which a further reduction is not sustainable to counter a decrease in the level of sustainability. Hence, they argue that substitution possibilities and uncertainty about future preferences (or production possibilities) are substitutes in the sense that an increase in the uncertainty about future preferences can be compensated by a lower elasticity of substitution and vice versa. As a result, more uncertainty provides a rationale for more preservation (Fisher and Krutilla, 1974).

Although the elasticity of substitution has significant implications with respect to future WTP values, risk aversion and, consequently, the discount rate used to estimate damages and benefits in present value terms (e.g. Ayong Le Kama and Schubert, 2004; Krysiak and Krysiak, 2006; Traeger, 2007), it is hard to provide good empirical estimates for the elasticity of substitution. As mentioned by Sterner and Persson (2008), the elasticity of substitution would vary considerably among environmental goods and services, as well as between individuals. In this case, it is expected that the relevant aggregate outcome will be dominated by goods and services with low elasticities (e.g. clean water). Finally, the elasticity of substitution is not a static measure as it may vary over time with scarcity.

### 3.1.4 Income elasticities

It is widely hypothesized that environmental quality is a ‘luxury good’ so that extra provision of environmental quality will benefit the rich more than the poor (McFadden, 1994). However, as Pearce (2003) mentions, this assumption has not been widely tested. In general, estimates of the income elasticity of demand are difficult to derive from non-market valuation studies because contexts with varying price and income combinations are seldom considered. This is the reason why what is usually observed is the income elasticity of WTP, which is estimated as follows:

$$\omega = \frac{\partial WTP}{\partial Y}\frac{Y}{WTP} \qquad \text{(Eq 28)}$$

where:

Y is income

WTP is the willingness-to-pay

The relationship between the income elasticity of demand and the income elasticity of WTP is not fixed, as an environmental good may have income elasticity of demand greater than 1 and income elasticity of WTP greater than or less than 1 (Flores and Carson, 1997). Therefore, the next sections provide estimates of both the income elasticity of demand and the income elasticity of WTP from relevant studies.

#### 3.1.4.1 Income elasticity of demand

Regarding the income elasticity of demand of environmental assets, the literature provides some evidence. Borcherding and Deacon (1972), using a logarithmic model of per capita expenditure, conducted a collective study to estimate the parameters that govern the demand of public services in the US for various public goods, including parks and recreation. The regression results show income elasticities between 0.5 and 2.7.

Bergstrom and Goodman (1973) developed a method for estimating demand functions of individuals for municipal public services by regressing the expenditures of municipalities on specific services, among them parks and recreation. According to the results, the income elasticity of demand is 1.32.

Thomas and Syme (1988) conducted a CV survey to estimate price and income elasticities of demand for public water supply. According to the econometric results estimated income elasticities are around +0.2.

Dalhuisan et al. (2003) conducted a meta-analysis of variations in price and income elasticities of residential water demand. The estimated income elasticities vary between -0.9 and +7.8, with a mean of 0.46 and a median of 0.28 (st. dev.=0.81). Approximately 10% of the estimates is greater than 1 and hence, again corroborating theoretical expectations, water demand appears to be inelastic in terms of income changes.

Hökby and Söderqvist (2003) using data from five Swedish CV studies provide estimates of income and price elasticities of demand for reduced marine eutrophication effects in the case of the Baltic Sea. The point estimate of the income elasticity of demand for the base case is 0.94, indicating that reduced eutrophication effects are a necessity. However, the 95% confidence interval ranges from 0.58 to 1.31 (i.e. a 1% increase [decrease] in income would result in about a 0.6–1.3% increase [decrease] in the demand for reduced eutrophication effects), which means that the necessity label is not statistically significant.

Worthington and Hoffman (2008) also provide a synoptic survey of empirical residential water demand analyses conducted in the last 25 years. They analyse both model specification and estimation and discuss the outcomes of the analyses. They estimate income elasticities, which are of a much smaller magnitude (0.01 to around 0.8 and positive, with one exception by Agthe and Billings (1980) who estimated short-run income elasticities between 1.33 and 2.07, and long-run income elasticities in the range of 1.97–2.77.

Khan (2009) used data from two studies and implemented survey-based data approach, similar to that of Hökby and Söderqvist (2003), for modelling the demand for improved environmental quality of two National Parks in Northern Pakistan and for estimating income and price elasticities. The point estimate of the income elasticity of demand for the base case is 1.10, indicating that improvement in park quality is a luxury good. Yet, the 95% confidence interval ranges from 0.71 to 1.49 (i.e. a 1% increase [decrease] in income would result in about a 0.7–1.5% increase [decrease] in the demand for improved quality), which means that the luxury label is not statistically significant.

Khan (2012) used a CV survey to ask a sample of respondents about their awareness and WTP for safe drinking water in Northern Pakistan Using the same approach, as in previous study, reports identical estimates for quality effects of drinking water. More specifically, the point estimate of the income elasticity of demand for the base case is 0.717.

Yoo et al. (2014) use a differenced regression model including direct measures of changes in water prices to estimate income elasticity of residential water demand in Phoenix, Arizona. Their (long-term) estimate of the income elasticity of water demand is about 0.036, showing that a 1% increase in median household income, between 2000 and 2008, increased water use by 0.036% over that interval. The fact that consumers with higher income tend to use more water is reported to be consistent with previous findings.

#### 3.1.4.2 Income elasticity of willingness-to-pay (WTP)

As mentioned, it is not as straightforward to define the income elasticity of demand for an environmental good as it is for the income elasticity of demand for a private good. In principle, the income elasticity of demand for private goods is different, provided that a household chooses private goods subject to a budget constraint, while an environmental good is exogenous to the household; therefore, the analysis of income elasticity in the case of non-priced public goods is quite different (Hanemann, 1991). Several empirical studies suggest that WTP is an increasing function of income (Kriström and Riera, 1996), i.e. respondents with more income tend to express higher WTP values for environmental improvements; that is, environmental improvements are typically normal goods. Yet, this is not always the case.

Kriström and Riera (1996) estimated the income elasticity of environmental improvements for six European CV datasets and they found that the value of the parameter is consistently less than one, with few exceptions. Horowitz and McConnell (2003) provided elasticities for 12 datasets and found that income elasticities of WTP in CV surveys range around 0.1–0.4.

Hökby and Söderqvist (2003) compiled 21 estimates of income elasticity of WTP from CV studies in Sweden. The point estimates vary between –0.71 and 2.83. Only one of the estimated elasticities is negative, and only four of 21 are greater than unity. The mean and median values of income elasticity of WTP are 0.68 and 0.46, respectively. Hence, they conclude that income elasticity of WTP tends to take values between 0 and 1, a finding consistent with the results reported by Kriström and Riera (1996).

Pearce (2003) discuss a general conceptual framework for the analysis of the socio-economic distribution of environmental costs and benefits using the available empirical literature. To this end, he provides evidence of the income elasticity of WTP from a number of studies, summarized in the following Table 2. Pearce (2003) points out that the income elasticity of WTP for environmental quality is, in general, less than 1. The exception to this basic rule is the findings by Sieg et al. (2000) who adopt an entirely different approach. In addition, the income elasticity of WTP for life risks produces is significantly higher than 1 when time series valuations are considered (i.e. in Costa and Kahn, 2002).

**Table 2.** Elasticities of WTP from non-market valuation studies

| **Study** | **Income elasticity of WTP** |
|---|---|
| Gianessi et al. 1977: compliance with 1970 Clean Air Act, USA | 0.35-0.87 |
| Harrison and Rubinfeld 1978: US clean air standards | 1.00 |
| Nelson 1978: noise in the US | 1.00 |
| Harris 1979: UK noise | 0.20-0.40 |
| Walters 1975: UK airport noise - Heathrow -Gatwick | 1.89-3.20<br>2.09-2.62 |
| Kriström and Riera, 1996 6 CVMs: Finland, France, Norway, Netherlands, Spain, Sweden | Probably less than 1 |
| Hökby and Söderqvist (2001). 21 estimated CVM equations in Swedish valuation studies | Range -0.71 to 2.83<br>Mean = 0.68<br>Median = 0.46 |
| Imber et al. 1991. Kakadu conservation zone, Australia | 0<br>0.20 |
| Carson et al. 1995. Exxon Valdez tanker spill in Alaska | 0.28 |
| Santos, 1998<br>Landscape change, UK<br>Landscape change, Portugal<br>Meta analysis of landscape studies | <br>0.20<br>0.30<br>0.57 |
| Loehman and De, 1982 Avoidance of respiratory symptoms, Florida | 0.26-0.60 |
| Jones-Lee et al. 1985 Accidents | 0.40 |
| Biddle and Zarkins, 1988<br>Occupational risk | 0.70 |
| Viscusi and Evans, 1990<br>Health status | 1.10 |
| Sieg et al, 2000<br>Air quality, S.California | 4.2-4.7 |
| Viscusi and Aldy, 2002<br>Life risks | 0.5-0.6 |
| Costa and Kahn, 2002<br>Life risks | 1.5-1.7 |

Source: Pearce, 2003

Jacobsen and Hanley (2009) conducted an empirical analysis using random effects panel models to examine the effects of income - and then GDP per capita - on WTP for habitat and biodiversity conservation. In their meta-analysis, 145 WTP estimates for biodiversity conservation where existence value plays a major role were collected from 46 CV studies across six continents. They found that income elasticity of WTP for biodiversity conservation is +0.38, both when using GDP per capita and household/personal income. Although the focus on this study was on existence values (i.e. non-use) and not on use values, it does not seem to change the conclusions from Kriström and Riera (1996) and Hökby and Söderqvist (2003) that WTP income elasticities lay between 0 and 1.

Ludwig and Neumann (2012) in a Memorandum to Office of Air and Radiation of the U.S. Environmental Protection Agency (US EPA), summarize income elasticity of WTP estimates for three types of health effects: minor, severe, and premature mortality, which are used in EPA's Benefits Mapping and Analysis Program (BenMAP), to adjust WTP values for avoided premature mortality and severe and minor morbidity from a variety of studies (Table 3).

**Table 3.** Recommended BenMAP income elasticities

| Health endpoint | Low estimate | Central estimate | High estimate |
|---|---|---|---|
| Minor Health Effects[a] | 0.06 | 0.30 | 0.70 |
| Severe and Chronic Health Effects[b] | 0.38 | 0.68 | 1.25 |
| Premature Mortality[c] | 0.20 | 0.65 | 1.44 |

Source: Ludwig and Neumann (2012)

Martini and Tiezzi (2013) conducted a demand analysis with household production to estimate the marginal WTP for improvements in air quality and the corresponding income elasticity of WTP in Italy. They calculated the income elasticity of WTP for the entire sample and for five income groups, estimating consistently positive income elasticities of WTP. The overall sample mean income elasticities of WTP is equal to 1.164 (range: 1.165 – 1.345), which suggests that, as societies get richer, they tend to value environmental quality more highly. Finally, they conclude that air quality improvements are not a luxury good, but a normal one. The demand for air quality improvements increases with income, and they are income neutral, because households are willing to spend a proportional share of their income as income grows.

# 4 Future economic choices with changing preference

The attitudes of future generations towards environmental goods and services may be different from those of present generations. This evolution/diversification of attitudes/preferences between generations can be considered both conspicuous and unpredictable simultaneously. It can be justified by the fact that preferences consist of completely different components and are triggered by a number of motives, which change significantly through the years. A change in preferences may occur in the future unexpectedly, affecting the level of utility associated with 'consumption' of environmental goods or services.

Generally, preferences can be considered that remain stable in the short-term future, but significant changes are expected in the long-term future (Skourtos et al 2010). To place the issue of value dynamics in the Millennium Ecosystem Assessment terminology, the temporal dimension of social benefits derived from ecosystem services vary from direct, short to medium term benefits (provisioning) to indirect, medium to long term benefits (regulating), to direct, long term benefits (cultural), to indirect, long to very long term benefits (supporting). The last category of long to very long temporal benefits is what some researchers would prefer to call ecological benefits in contrast to the short to medium term socio-economic benefits. To this direction, strong indications have been identified that future generations will tend to be more sensitive about the environment (greening of preferences) leading to higher WTP values. Indisputably, the deterioration of the quality of environmental assets in combination with the upcoming depletion of natural resources constitutes the main reason for this tendency. The values of environmental goods and services are affected mainly by the triggered impacts in the long-term future confirming the assumption that future preferences will most probably be different in comparison with existing ones.

Nevertheless the opposite could also be true: according to Ayong Le Kama and Schubert (2004) future generations may value environmental goods and services less in comparison with existing generations. The authors allude that the relation of consumption to the levels of welfare is critical for the validation of this assumption. Specifically, the authors analyse specific changes in future preferences, which could lead to a more conservative use of natural resources with the prerequisite that the environmental quality is taken into consideration. Therefore, it is vital to separate the impacts of consumption and environmental quality in the utility function in order to identify the evolution of future preferences. Furthermore, the authors suggest selecting the reference scenario regarding changes in preferences in the short-term future and becoming more conservative in the long-term future taking into account specific conditions such as the structure of the economy and the probability of a change in the preferences of the people.

Two different behaviours can be expected due to potential changes in preferences. Firstly, if the growth of economy and the environment quality are both considerably low, people will tend to be more provident and conservative. However, in case that the growth of economy is high and the environmental quality fair, the people will increase the existing levels of consumption despite the deterioration of the environmental quality adopting a type of insurance behaviour against the future conditions. As a result, the estimation of the changes in the preferences depends on the concern about the well-being of future people requiring the estimation of both the living conditions and the corresponding preferences in the future. Therefore, assumptions about the impact of the present actions on the future living conditions and the respective changes of the preferences are required for the analysis of changing preferences (Krysiak and Krysiak, 2006).

According to Krysiak and Krysiak (2006), future preferences cannot be taken into consideration nowadays, due to the fact that the respective individuals are not yet present. The present generations have specific expectations about future preferences, while the actual future preferences will probably be different from these expectations. As a result, the estimation of future preferences is characterized by a high degree of uncertainty; the authors propose a specific model within the context of sustainability for the effective manipulation of this uncertainty.

Besides the identified uncertainties, materialism may result in lower WTP values owing to less concern about the effects of environmental damage on other people, ecosystem, and future generations. Materialism constitutes a value orientation, which influences the environmentalism at a significant degree (Kidd & Lee, 1997; Kilbourne & Pickett, 2008).

Hultman et al (2015) developed a conceptual model in order to examine the environmental beliefs, attitudes toward ecotourism, behavioural intentions and WTP premium in relation with materialistic and tourist motivation. Specifically, a survey was conducted with Swedish and Taiwanese tourists in order to test the established model. A positive relation between environmental attitudes and willingness to pay and a negative relation between materialism and willingness to pay were the main findings from the performed analysis. Moreover, a negative relation was identified between materialism and environmental beliefs. An elliptical re-weighted least squares method was applied in order to test the hypothesized direct relationships. The results of these models are presented in Table 4.

**Table 4.** Relation of materialism with WTP and environmental beliefs

| **Examined variables** | **Expected sign** | **Swedish tourists** | | **Taiwanese tourists** | |
|---|---|---|---|---|---|
| | | **β** | **t** | **β** | **t** |
| Materialism vs Environmental beliefs | - | -0.13 | -2.44* | -0.07 | -1.09 |
| Materialism vs WTPP | - | -0.17 | -4.12** | -0.14 | -3.03** |

Source: Hultman et al. (2015)

Further, Inglehart's (1981) post-materialistic value approach has led to the identification of a divergent relationship between materialism and environmentalism. Post-materialism presumes that economic insecurity during the pre-adulthood years has as a result the formulation of materialistic values and the prioritization of lower-order needs in adulthood. In contrast, economic security leads to lower materialistic values and to the prioritization of higher-order values, such as in the cases of environmental goods and services (Davis 2000).

Focusing on the effect of post-materialism, Goksen et al. (2002) analyse the impact of geographical proximity of environmental problems on the formulation of environmental beliefs and willingness-to-pay in a scenario of interventions for environmental improvements. A survey was conducted in Turkey in order to measure WTP for three different environmental problems, namely sea pollution, soil erosion and ozone depletion at local, national and global level correspondingly. According to the obtained results the existence of post-materialism leads to more environmental concern and to higher amounts of WTP for the protection of the environment at local and global levels. The results of the analysis are presented in Table 5.

Finally, consumerism can also affect and shape preferences of future generations. Specifically, Yu et al (2014) examined the WTP of citizens for the purchase of ‘‘green food’’ in China. According to the results, a positive relation was identified between WTP and the frequency of shopping indicating that a potential increase of consumerism in future will lead to higher WTP values.

**Table 5.** Relation of postmaterialism and other parameters with WTP

| | WTP for sea pollution in Istanbul (local issues n=524) | WTP for soil erosion in Turkey (national issues n=524 | WTP for ozone depletion (global issues n=517 |
|---|---|---|---|
| Education | 0.911 (0.048) | 1.014 (0.048) | 1.003 (0.045) |
| Urbanity | 0.941 (0.051) | 0.949 (0.049) | 1.003 (0.054) |
| Postmaterialism | 0.942* (0.029) | 0.987 (0.026) | 0.904** (0.024) |
| Material security | 0.969 (0.018) | 0.941** (0.017) | 0.946** (0.015) |
| Log likelihood | -646.619 | -676.681 | -680.371 |
| $\chi^2$ | 27.62 | 18.90 | 38.44 |
| Probability > $\chi^2$ | 0.000 | 0.000 | 0.000 |

Source: Goksen et al (2002).

# 5 Modelling future economic choices with fixed and evolving preferences

## 5.1 Factors affecting future willingness-to-pay values

Based on prior sections, it becomes apparent that the evolution of WTP values will be influenced mainly by the following factors:

- The growth of income ('income growth factor')
- The depletion of environmental assets ('depletion factor')
- The elasticity of substitution between man-made and environmental goods and services ('substitution factor');
- The change in preferences of future generations ('changing preferences factor')

In what follows though, the substitution factor is omitted in our model in order to avoid double-counting issues since:

- In case of perfect substitution between man-made and environmental assets the effect of scarcity on WTP values is negligible and the nonmarket damages are estimated as consumption losses (see the discussion in Section 3.2);
- In case of weak substitution between man-made and environmental assets the effect on WTP values is captured via the depletion factor as discussed hereinafter.

To take into account the effect of income growth factor ($\alpha_{inc}$) it is necessary to consider both the income elasticity of WTP and the growth rate of income, i.e.

$$\alpha_{inc} = \omega . g \qquad \text{(Eq 29)}$$

where:

g is income or growth rate on an annual basis and expressed as percentage increase (or decrease) from previous year

ω is the income elasticity of WTP, that is:

$$\omega = \frac{\Delta WTP\%}{\Delta Inc\%} = \frac{dWTP}{WTP}\frac{Inc}{dInc} \quad \text{(Eq 30)}$$

The environmental depletion (or scarcity) factor ($\alpha_{sc}$) is given by:

$$\alpha_{sc} = \lambda q \quad \text{(Eq 31)}$$

where:

q is the depletion rate on an annual basis and expressed as percentage decrease (or increase) from previous year

λ is the 'price' (i.e. WTP) elasticity of demand, estimated as follows:

$$\lambda = \frac{\Delta WTP\%}{\Delta Q\%} = \frac{dWTP}{WTP}\frac{q}{dQ} \quad \text{(Eq 32)}$$

Alternatively and assuming that the environmental resource is exhaustible, the scarcity factor, is equal to the social discount rate s in the spirit of the Hotelling rule, i.e.

$$\alpha_{sc} = s \quad \text{(Eq 33)}$$

where s is the social discount rate

Finally, an evolution of preferences may take place in the future and may modify (in an unknown way) the attitude of future generations towards environmental assets. As mentioned in Section 4, some of the factors associated with these changes are obvious but others are not. Therefore, the formation of future preferences involves complex and interlinked economic, social and moral determinants. For all these reasons, it is not possible to foretell whether people will care more about the environment in the future (i.e. green preferences), or less (i.e. materialistic preferences). Since future preferences are unknown, the preferences factor (αpr) works in our model as a 'drift' (upwards or downwards) and is defined ad hoc in a range of possible numerical values between pure 'green' and 'materialistic' preferences (see Fig. 3).

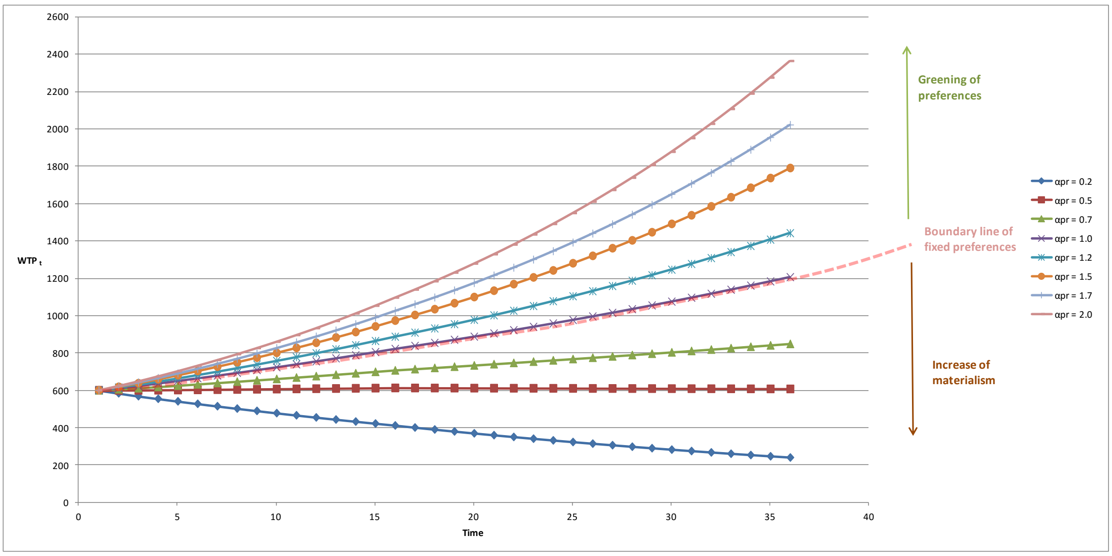



**Fig. 3.** The effect of $a_{pr}$ on future WTP values

## 5.2 Model estimation

Given that each and every of the above-mentioned factors has an independent and additive effect on the evolution of future WTP values, the total growth rate, $\alpha_{tot}$, of WTP is given by:

$$\frac{W\dot{T}P}{WTP} = \omega\frac{In\dot{c}}{Inc} + \lambda\frac{\dot{Q}}{Q} + \alpha_{pr} \qquad \text{(Eq 34)}$$

When the right-hand expression of equation 34 is constant, the solution is $WTP_t = WTP_0e^{\alpha t}$ with α given by the right-hand side of equation 45 below:

$$\alpha_{tot} = \alpha_{inc} + \alpha_{sc} + \alpha_{pr} \qquad \text{(Eq 35)}$$

where:

$\alpha_{tot}$ is the total growth rate of WTP

$\alpha_{inc}$ is the income growth factor

$\alpha_{sc}$ is the environmental depletion (or scarcity) factor

$\alpha_{pr}$ is the changing preferences factor

It is evident that the aggregate growth rate $a_{tot}$ is quite conjectural and therefore likely to change over time in an unspecified way. Consequently, the numerical value of $a_{tot}$ can never be ex ante determined faithfully. This is a situation with many similarities to those observed in financial and economic applications, e.g. the prediction of stock prices. Generally speaking, there are two approaches commonly used to predict future outcomes in this field, namely: (a) structural approaches, which are based on the theory of fundamental or intrinsic value analysis, and (b) time-series behaviour models, which rely on “chartist” or “technical” theories (Fama, 1965; Newell and Pizer, 2003). The structural approach assumes that the parameter under investigation is as an endogenous outcome of a dynamic general equilibrium model, which depends on some exogenous fundamental factors. A careful study of the fundamental factors may allow the analyst to determine whether the value of the parameter is above or below its intrinsic value, i.e. the parameter is estimated by examining the relationships among the variables. Technical theories assume that past patterns of the parameter investigated will tend to recur in the future and, thus, establish a transparent connection between historic data and forecast values.

According to the literature review presented in previous sections, elasticities of income and demand vary significantly. Furthermore, since our focus is in the very-long-term, it is not possible to predict the magnitude and timing of future events and preferences that might influence the growth rate $\alpha_{tot}$. Therefore, simulation functions were constructed using flexible random walk-based stochastic models with and without drift (i.e. with and without change of preferences) to deal with the uncertainties involved. Random walk models are used in finance and economics, e.g. in understanding and predicting the behaviour of stock market prices (e.g. Fama, 1965; Smith, 2002; Borges, 2011), interest rates (e.g. Pesando, 1979; Newell and Pizer, 2003; Bacchetta and Van Wincoop, 2007), growth and inequality (e.g. Scalas, 2006), etc.

The random walk-based stochastic process is equivalent to a Brownian motion. Formally, random walk is a Wiener process W with domain $(0,\infty]$, named after Norbert Wiener’s work in the early 1920s, such that (Shafer and Vovk, 2001) W(0) for each $t>0$, W(t) is Gaussian with mean zero and variance t. If the intervals $[t_1,t_2]$ and $[u_1,u_2]$ do not overlap, then the random variables $W(t_2)-W(t_1)$ and $W(u_2)-W(u_1)$ are independent. If $d_t$ is a small positive number, then the increment $W(t+d_t)-W(t)$ is Gaussian with mean zero and variance $d_t$. In practice, a Brownian motion is any process S of the form:

$$S(t) = \mu_t + \sigma W(t) \qquad \text{(Eq 36)}$$

with $\mu \in R$ and $\sigma \geq 0$.

In this case, S(t) is a random variable with mean $\mu_t$ and variance $\sigma^2_t$, where μ is the drift of the process and σ its volatility. Hence, for any positive real number $d_t$ equation (36) becomes:

$$dS(t) = \mu d_t + \sigma dW(t) \qquad \text{(Eq 37)}$$

where W(t) is a Wiener process with μ and σ constants. As noted, μ controls the trend and σdW(t) controls the “random noise” effect.

Based on the general model, the aggregate growth rate α of WTP in period (t) is estimated as follows:

$\alpha_{tot}(t) = \theta(0) + k_t + \varepsilon_t$ (Eq 38)

where:

$\alpha_{tot}(t)$ is the total growth rate of WTP at time t

$\theta(0)$ is the sum of $\alpha_{inc} + \alpha_{sc}$ at time 0

k is the drift that reflects changes in future preferences

$\varepsilon_t$ is a random component estimated by $\sigma W(t)$

Since the growth rate of WTP is a continuous process dependent on its rate of change and time, in order to properly characterize it we should express $\alpha_{tot}(t)$ as dependent on the differential change of the rate, i.e. it has to be rewritten as a differential process:

$d\alpha_{tot}(t) = kd_t + \sigma dW(t)$ (Eq 39)

Hence, $d\alpha_{tot}(t)$ is normally distributed with mean $k_t$ and standard deviation σ. Thus, in order to implement the model the following parameters should be quantified:

- The income elasticity of WTP (ω)
- The income or growth rate (g) on an annual basis
- The WTP elasticity of demand (λ)
- The environmental depletion rate (q) on an annual basis
- The preferences factor ($\alpha_{pr}$), which is expressed by k
- The volatility (σ) for the stochastic term

The elasticities ω and λ can be obtained from existing nonmarket valuation studies (see Sections 3.1 and 3.3); the preferences factor is defined ad hoc (i.e. it depends on the future preference scenario chosen by the analyst). To facilitate this selection, it is suggested to express the evolution of preferences using the ratio $WTP_{2050}/WTP_{2015}$, i.e. to make an arbitrary selection about the future WTP value. Having defined $WTP_{2015}$ and $WTP_{2050}$, the average preferences factor is estimated as follows:

$k = \frac{1}{35} ln \frac{WTP_{2050}}{WTP_{2015}}$ (Eq 40)

Similarly, in order to calculate the annual environmental depletion rate q, the analyst may use the ratio $\Delta Q = Q_{2050}/Q_{2015}$ which expresses the expected decrease (or increase) in the quality or quantity of the environmental good or service under investigation, according to the following equation:

$q = \frac{1}{35} ln \frac{Q_{2050}}{Q_{2015}}$ (Eq 41)

Finally, the volatility σ is usually estimated in similar models using historical data series (see for example Newell and Pizer 2003). Nevertheless, in our model this is not possible owing to data unavailability. Therefore, a different approach is implemented by means of a Monte Carlo simulation of the growth rate $\alpha_{tot}$. The Monte Carlo simulation is described in Section 6. In Fig. 4 and 5 ten random walks of the model for the total growth rate $\alpha_{tot}$ and WTP are presented for the time period 2015-2050 assuming a today's WTP of €100 and a total growth rate $\alpha_{tot}$ equal to 4.38%.

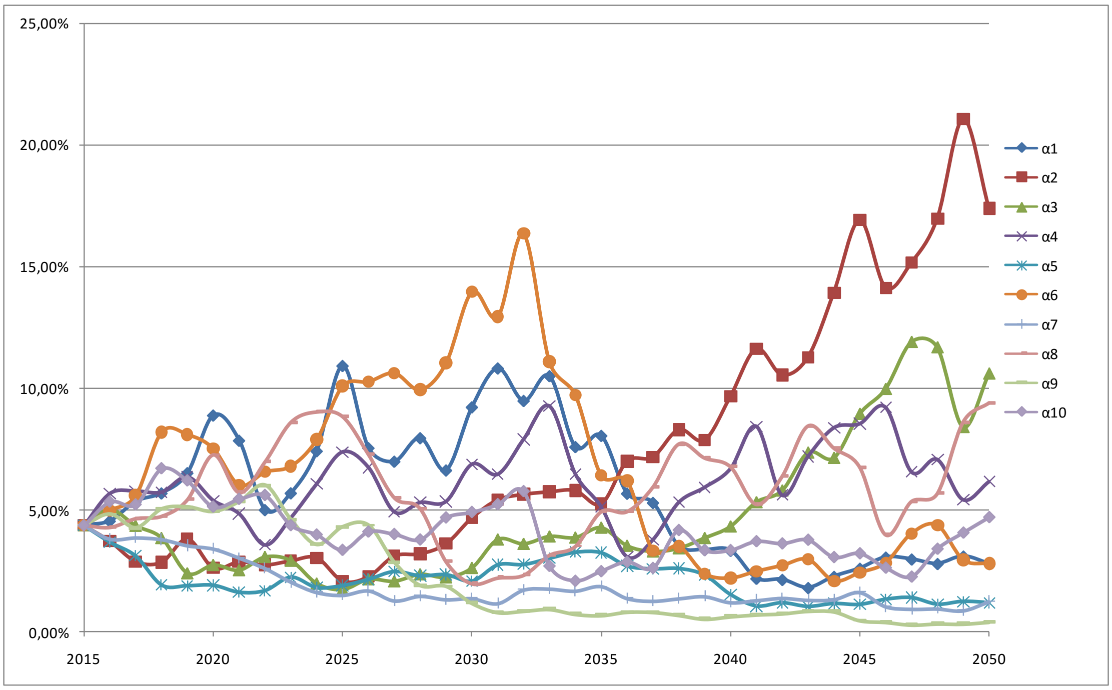



**Fig. 4.** Random walks of $\alpha_{tot}$ for the period 2015-2050

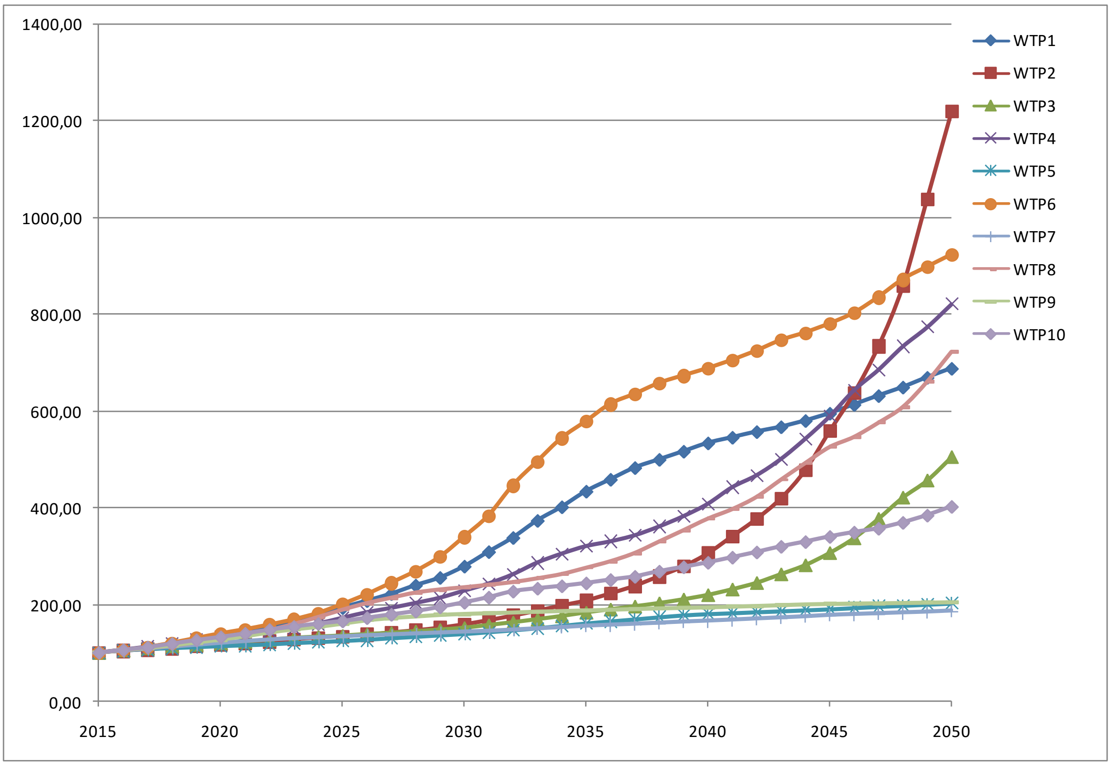



**Fig. 5.** Random walks of WTP value for the period 2015-2050

# 6 An illustrative application of the model

## 6.1 Case study

For illustrative purposes, we choose to estimate the effect of growth rate $\alpha_{tot}$ to the WTP values of Temperate and Boreal forests (i.e. temperate deciduous forests, temperate broadleaf and mixed forests, temperate coniferous forests, and temperate rainforests) that constitute a significant part of the total forest area in Europe. To this end, the results of the Ecosystem Service Value Database are used, as presented by de Groot et al. (2012). An overview of monetary values per ecosystem service of forests is given in Table 6. In addition, the list of the monetary values used for these calculations as provided by de Groot et al. (2012), is given in the Appendix.

From the above-mentioned services provided by temperate and boreal forests our illustrative analysis focuses on the recreational opportunities. To this end, the parameters of the model (e.g. the WTP elasticity of demand) are selected accordingly.

**Table 6.** Monetary value of services provided by Temperate and Boreal Forests (in Int. $/ha/year-2007 values)

| | No. of estimates | Mean value | Median value | St.Dev. | Min value | Max value |
|---|---|---|---|---|---|---|
| ***Provisioning services*** | ***9*** | ***671*** | ***450*** | ***867*** | ***121*** | ***1593*** |
| 1 Food | 2 | 299 | 299 | 422 | 0 | 597 |
| 2 Water | 3 | 191 | 121 | 123 | 118 | 333 |
| 3 Raw materials | 4 | 181 | 31 | 322 | 2 | 662 |
| 4 Genetic resources | | | | | | |
| 5 Medicinal resources | | | | | | |
| 6 Ornamental resources | | | | | | |
| ***Regulating services*** | ***13*** | ***491*** | ***367*** | ***584*** | ***105*** | ***1212*** |
| 7 Air quality regulation | | | | | | |
| 8 Climate regulation | 6 | 152 | 34 | 241 | 7 | 624 |
| 9 Disturbance moderation | | | | | | |
| 10 Regulation of water flows | | | | | | |
| 11 Waste treatment | 3 | 7 | 0 | 13 | 0 | 22 |
| 12Erosion prevention | 1 | 5 | 5 | | 5 | 5 |
| 13 Nutrient cycling | 1 | 93 | 93 | | 93 | 93 |
| 14 Pollination | | | | | | |
| 15 Biological control | 2 | 235 | 235 | 330 | 1 | 469 |
| ***Habitat services*** | ***10*** | ***862*** | ***171*** | ***1342*** | ***51*** | ***3573*** |
| 16 Nursery service | | | | | | |
| 17 Genetic diversity | 10 | 862 | 171 | 1342 | 51 | 3573 |
| ***Cultural services*** | ***26*** | ***990*** | ***139*** | ***2644*** | ***1*** | ***10028*** |
| 18 Aesthetic information | | | | | | |
| 19 Recreation | 25 | 989 | 138 | 2644 | 1 | 10027 |
| 20 Inspiration | | | | | | |
| 21 Spiritual experience | | | | | | |

| | | | | | | |
|---|---|---|---|---|---|---|
| 22 Cognitive development | 1 | 1 | 1 | | 1 | 1 |
| **TOTAL** | **58** | **3013** | **1127** | **5437** | **278** | **16406** |

Source: de Groot et al. (2012)

## 6.2 Estimating model parameters

### 6.2.1 Present WTP value

According to the estimates presented in Table 6, the mean value of forestland for recreation is 989 Int.$ per ha and per year, ranging from 1 Int.$ per ha and per year up to 10,027 Int.$ per ha and per year (all values in 2007 prices). In order to offset influences concerning differences of income, price level and time, we express the original values to Euros in 2015 prices using the methodology for benefit transfer proposed by Pattanayak et al. (2002):

$$WTP_{p,pt} = WTP_{s,st} * \frac{PPPI_{p,st}}{PPPI_{s,st}} * \frac{CPI_{p,pt}}{CPI_{p,st}} \quad \text{(Eq 42)}$$

where:

p denotes the policy site and s the study site

$p_t$ refers to the year that the benefit transfer study is conducted and $s_t$ refers to the year that the original study was conducted

PPPI is the Purchasing Power Parity Index

CPI is the Consumer Pricing Index

Using the PPPI and CPI values obtained by the World Bank (2015) and OECD (2015), the mean, minimum and maximum values per ha and per year in Euros2015 are 922, 0.9 and 9,345, respectively.

### 6.2.2 Income elasticity of WTP

Based on the results of the literature review, a central (i.e. likeliest) estimate of 0.7 was adopted for the income elasticity of WTP. In addition, for conducting sensitivity and probabilistic analyses the minimum and maximum values were taken equal to 0.1 and 1, correspondingly.

### 6.2.3 Annual growth rate

In order to estimate the annual growth rate, estimates from international organizations were taken into consideration. According to OECD (2012a), the growth of the present non-OECD economies will continue to outpace that of the present OECD countries in the coming decades. However, from over 7% per year on average over the last decade, non-OECD growth may decline to around 5% in the 2020s and to about half that by the 2040s. As regards the OECD economies, the average annual growth rate is estimated between 1.9% and 2.2% (Table 7).

**Table 7.** Growth in total economy potential output

| Potential real GDP growth (%) | 2010 | 2012-2017 | 2018-2030 | 2031-2050 |
|---|---|---|---|---|
| OECD | 1.5 | 2.0 | 2.2 | 1.9 |
| non-OECD | 7.5 | 6.9 | 5.1 | 3.0 |
| World | 2.7 | 3.4 | 3.3 | 2.4 |

Source: OECD (2012a)

According to PwC (2015) the E7 economies (i.e. China, India, Brazil, Russia, Indonesia, Mexico and Turkey) will continue to be the driving force of the world economy with an annual average rate of growth 3.8% during the period 2014 – 2050. The G7 (i.e. US, Japan, Germany, UK, France, Italy and Canada) economies are expected to grow at an average rate of 2.1% per annum over the same time period (Fig. 6).

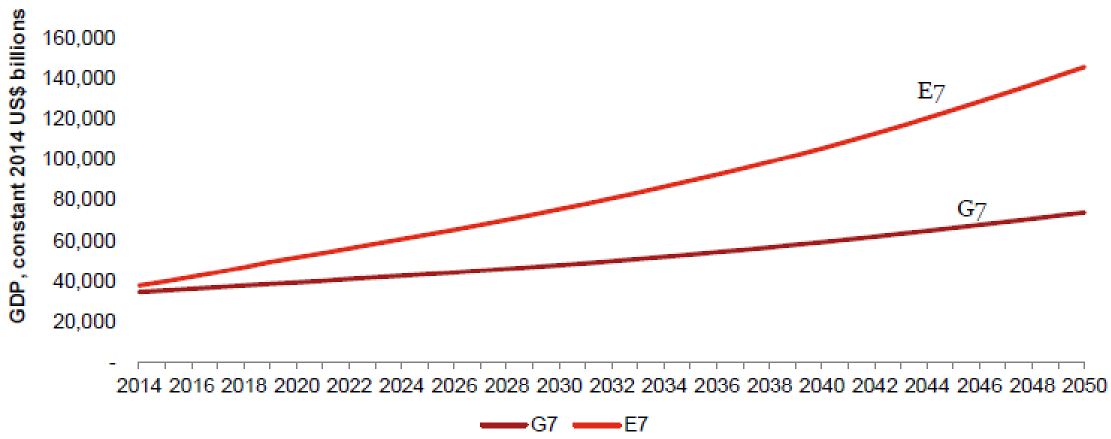



**Fig. 6.** Projected growth of GDP for E7 and G7 countries 2014-2050 (Source: PwC, 2015)

For purposes of our illustrative example, our model assumes an average rate of 2.0% per annum for European economies.

### 6.2.4 WTP elasticity of demand

Concerning the ‘environmental’ elasticity of WTP, an average unitary elasticity (-1) was considered as the base value for the estimates. Furthermore, a minimum value of -2.3 and a maximum value of -0.5 were considered for sensitivity purposes.

### 6.2.5 Environmental depletion rate

The estimation of the environmental depletion rate was performed taking into consideration the main findings from an OECD study on the evolution of the biodiversity until 2050 (OECD, 2012b). The projection for the case of the mature forests (primary forest) foresees a steady decrease until 2050 in all the examined regions within the framework of the baseline scenario (Figure 7). Specifically, the reduction of the primary forest area in OECD countries is expected to equal approximately to 14%. The corresponding reductions for the BRIICS countries, for the remaining countries and for all the countries at global scale will be 11%, 20% and 13%.

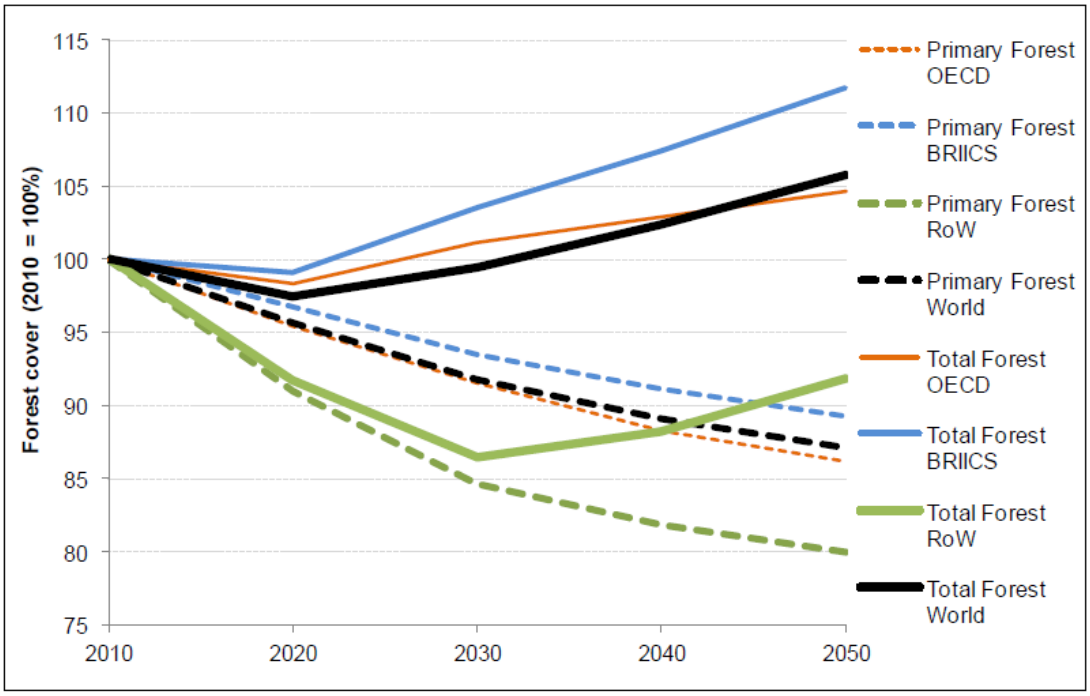



**Fig. 7.** Projections for changes in the forest area (Source: OECD, 2012b)

### 6.2.6 Preferences factor

Provided that future preferences are unknown, the preferences factor, αpr is defined ad hoc for different behavioural partners (scenarios). More specifically, three different behavioural scenarios were examined, as follows:

- Scenario A: Stable preferences. In this case the ratio WTP2050/WTP2015 equals to 1 and, consequently, the average (annual) preferences factor is zero.
- Scenario B: Green preferences. In this case the ratio WTP2050/WTP2015 is assumed to be 2 and, consequently, the annual preferences factor is 1.98%.
- Scenario C: Materialistic preferences. In this case the ratio WTP2050/WTP2015 is assumed to be 0.5 and, consequently, the annual preferences factor is -1.98%.

### 6.2.7 Volatility

As mentioned above, the volatility factor σ is a challenging issue contrary to the volatility of financial options, which is based on available historical information on the value of the stock examined (Kontogianni et al., 2013). There are several methods to estimate the volatility factor, e.g. the logarithmic cash flow returns, the project proxy approach, etc. In our case, the management assumption approach was employed using the results of the Monte Carlo simulation for the total growth rate αtot. To this end, we constructed a numerical approximation to the total growth rate αtot including uncertainty about the income elasticity of WTP, the price elasticity of demand and the environmental depletion rate. In total, 1,000 possible future growth rate αtot values were estimated starting in 2015 and extending 35 years into the future, i.e. up to 2050.

In order to represent the wider range of uncertainty that affects future growth rate αtot, the Maximum Entropy approach was chosen. As Gay and Estrada (2010) note, Maximum Entropy Principle is "…a useful tool for constructing probabilistic climate change scenarios that are the least biased estimates possible, consistent with the information at hand (including expert or decision-maker judgment) and that maximize what is not known…".

The idea behind maximum entropy is to formulate a distribution for the data such that the distribution maximizes the uncertainty in the data, subject to known constraints (Meyer and Booker, 2001). This definition of entropy, introduced by Shannon (1948), resembles a formula for a thermodynamic notion of entropy. For a continuous probability density function p(x) on an interval I, its entropy is defined as:

$$h(p) = -\int_I \; p(x) \ln p(x)\, dx \qquad \text{(Eq 43)}$$

Using Shannon's entropy measure, Jaynes (1957) showed that the maximum entropy estimate is the least biased estimate possible on the information at hand and it maximizes the uncertainty subject to the partial information that is given. This means that the choice of any other distribution will require making additional assumptions unsupported by the given constraints (Duracz, 2006). A direct derivation of the maximum entropy distribution involves solving a system of nonlinear equations, the solution of which involves variational calculus using the Lagrange multiplier method. The maximum entropy distribution can help assign probability distributions given certain constraints. For instance, when only the lower and upper bounds for an uncertain parameter are known, the principle of maximum entropy would indicate a uniform distribution. When the minimum, maximum and mode values are given, the triangular distribution that maximizes the entropy is chosen (Harr, 1987, quoted in Mishra, 2002). Regarding the uniform distribution, the minimum and maximum values for defining the minimum and maximum values of the distribution are used, as follows:

$$U(a.b) = [\min \alpha_L^{\mathrm{i}} , max\alpha_U^{\mathrm{i}}] \qquad \text{(Eq 44)}$$

where:

$\min \alpha_{\mathrm{L}}^{\mathrm{i}}$ is the minimum of the minimum values elicited by the literature review

$\max\alpha_U^i$ is the maximum of the maximum values elicited by the literature review

The triangular distribution is defined according to the following equation:

$$A = (a, b, c) = [\min \alpha_L^i, \frac{1}{n}\sum_{i=1}^{n} a_m^i, \max \alpha_U^i] \quad \text{(Eq 45)}$$

where:

$\min \alpha_L^i$ is the minimum of the minimum values elicited by the literature review

$\max\alpha_U^i$ is the maximum of the maximum values elicited by the literature review

$\frac{1}{n}\sum_{i=1}^{n} a_m^i$ is the average of central values elicited by the literature review

In order to better represent the uncertainty involved in WTP elasticities, the minimum, maximum and central values provided by the literature were combined in a triangular distribution, as follows:

Income elasticity of WTP: min = 0,1; likeliest = 0.7; max = 3.0

WTP elasticity of demand: min = -3.4; likeliest = -0.7; max = 0.0

As regards environmental depletion, a uniform distribution was adopted for the ratio $\Delta Q$ = Q2050/Q2015 with min and max values equal to 70% and 110%, respectively. The simulation results are given in Tables 7 and 8.

**Table 8.** Monte Carlo simulation summary statistics for the total growth rate $\alpha_{tot}$

| **Statistics** | **Forecast value** |
|---|---|
| Mean | 5.53% |
| Median | 5.39% |
| Standard Deviation | 1.12% |
| Minimum | 3.14% |
| Maximum | 9.37% |
| Mean Std. Error | 0.04% |

**Table 9.** Monte Carlo simulation percantiles for the total growth rate $\alpha_{tot}$

| **Percentiles** | **Forecast values** |
|---|---|
| 100% | 3.14% |
| 90% | 4.16% |
| 80% | 4.50% |
| 70% | 4.84% |
| 60% | 5.14% |
| 50% | 5.39% |
| 40% | 5.77% |
| 30% | 6.08% |

| | |
|---|---|
| 20% | 6.45% |
| 10% | 6.95% |
| 0% | 9.37% |

Using the results of Monte Carlo simulation and assuming that the total growth rate $\alpha_{tot}$ fluctuates within a normal distribution, the implied volatility can be calculated as follows (Mun, 2006):

$$\sigma = \frac{\alpha_{tot} percantilevalue - mean\alpha_{tot}}{mean\alpha_{tot} x inverse of the percantilevalue} \quad \text{(Eq 46)}$$

In this case, the volatility σ is about 20%. It is worth noting that the volatility factor used should be consistent with the time step adopted in the corresponding equations (Kodukula and Papudesu, 2006).

## 6.3 Present value of forest with fixed WTP

As a first step towards highlighting the effect of growth rate αtot on future WTP values and, consequently, on the estimated present value (PV) of benefits or costs, the PV of recreational value of 1 ha of forest land for the period 2015 – 2050 is estimated. To this direction, the original (i.e. today's) value remains constant (i.e. the estimates are in constant prices) and equal to 922€ (2015). This value is discounted at an appropriate social rate.

The debate in the literature regarding the appropriate social discount rate is reflected in the divergence of approaches in practice. As mentioned by Scarborough (2010), the Australian government recommends a social discount rate of 7% (with sensitivity analysis at 3% and 11%) for policy appraisal, while the UK Treasury suggests a rate of 3.5 per cent (H.M. Treasury, 2003). The disparity in the estimates derives from the assumptions made when implementing the "Ramsey" rate, as illustrated in Table 10.

**Table 10.** Examples of social discount rate estimates

| | Pure rate of time preference (δ)(per cent per annum) | Marginal elasticity of utility (η) | Rate of growth in consumption (g) (per cent per annum) | Social discount rate(r) (per cent per annum) |
|---|---|---|---|---|
| Nordhaus (2007) | 1.5 | 2 | 2 | 5.5 |
| Stern (2007) | 0.1 | 1 | 1.3 | 1.4 |
| Weitzman (2007) | 2 | 2 | 2 | 6 |
| UK Treasury Green Book [0-30 years] (2003) | 1.5 | 1 | 2 | 3.5 |

Source: Scarborough (2010)

An extensive discussion around the selection of the social discount rate is beyond the scope of this report. For illustrative purposes, a numerical value 3.5% for s is adopted as suggested by UK Treasury (H.M. Treasury, 2003). The PV of annual WTP recreational values per ha of forestland is estimated according to the following equation:

$$PV_{tot} = \sum_{t=0}^{35} \frac{WTP_t}{(1+s)^t} \qquad \text{(Eq 47)}$$

Where:

$PV_{tot}$ is the total present value of WTP for recreation per ha of forestland

$WTP_t$ is the annual WTP for recreation per ha of forestland at time t

s is the social discount rate

t is the time elapsed in years

Using the central estimate of 922 €2015, the $PV_{tot}$ for recreation per ha of forestland is estimated at approximately 19,360 €(2015).

Furthermore, in order to quantify variability due to the range of exiting WTP estimates, a typical approach to probabilistic modelling, i.e. Monte Carlo simulation is used. In our Monte Carlo simulation a model is run repeatedly 1,000 times using today's WTP as input parameter. Each time a different value for today's WTP is randomly generated based on a triangular probability distribution for the parameter. The triangular probability distribution is constructed according to the Maximum Entropy approach. To this end, the likeliest, minimum and maximum WTP values used are 922, 0.9 and 9,345 Euros2015 per ha and per year, respectively. The results of the Monte Carlo simulation are presented in Tables 11 and 12.

**Table 11.** Monte Carlo simulation summary statistics for the PV of WTP ignoring the total growth rate $\alpha_{tot}$

| **Statistics** | **Forecast value** |
|---|---|
| Mean | 72198.42 |
| Median | 64482.86 |
| Standard Deviation | 44319.00 |
| Minimum | 1546.88 |
| Maximum | 194610.81 |
| Mean Std. Error | 1401.49 |

**Table 12.** Monte Carlo simulation percentiles for the PV of WTP ignoring the total growth rate $\alpha_{tot}$

| **Percentiles** | **Forecast values** |
|---|---|
| 100% | 1546.88 |
| 90% | 18735.76 |
| 80% | 29945.29 |
| 70% | 40531.03 |
| 60% | 52469.76 |
| 50% | 64465.77 |

| | |
|---|---|
| 40% | 78705.21 |
| 30% | 95218.98 |
| 20% | 113398.08 |
| 10% | 136683.11 |
| 0% | 194610.81 |

According to the probabilistic simulation, the mean PV of WTP, ignoring the growth rate $\alpha_{tot}$, is around 72,200 Euros (2015), ranging between 1,550 and 194,600 Euros(2015).

## 6.4 Present value of forest with changing WTP

### 6.4.1 Scenario A: Stable preferences

According to Scenario A, the preferences remain unchanged during the time period considered (i.e. 2015 – 2050). Nevertheless, WTP values are influenced by the growth of income and the decrease in the area of primary forests. The former is captured in the model by $\alpha_{inc}$ and the latter is expressed by $\alpha_{sc}$. In this particular Scenario, $\alpha_{pr}$ is zero. The values used in the base-case estimates are: WTP2015 = 922 €; $\alpha_{inc}$ = 1.40% (which is estimated assuming an average growth rate of 2% per year and income elasticity of 0.7); $\alpha_{sc}$ =0.46% (which is estimated assuming an average depletion rate of 0.46% per year and 'environmental' elasticity of WTP of -1); and s =3.5% (i.e. the social discount rate).

Given that the model is based on a stochastic random walk process, 100 repetitions were made to estimate the mean PV. It is noted that in this case - contrary to the Monte Carlo simulation - the values of WTP, $\alpha_{inc}$ and $\alpha_{sc}$ remain constant. Thus, the results are only affected by the random component of the model, which is estimated by $\sigma W(t)$. The following Figures 8 and 9 illustrate ten of the random walks conducted during the modelling process.

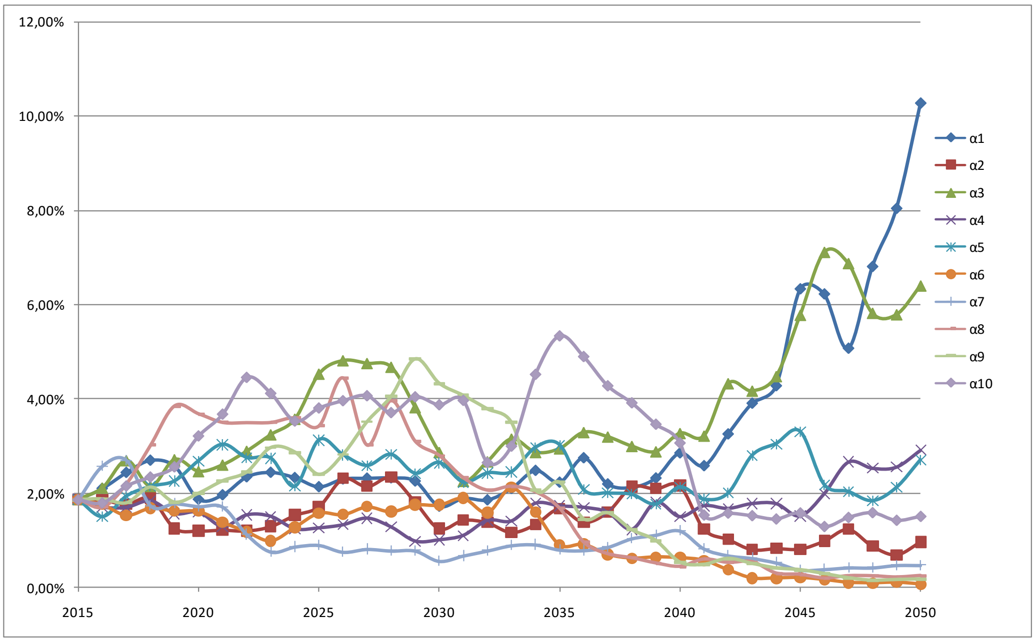



**Fig. 8.** Random walks of $\alpha_{tot}$ for the period 2015-2050, for Scenario A

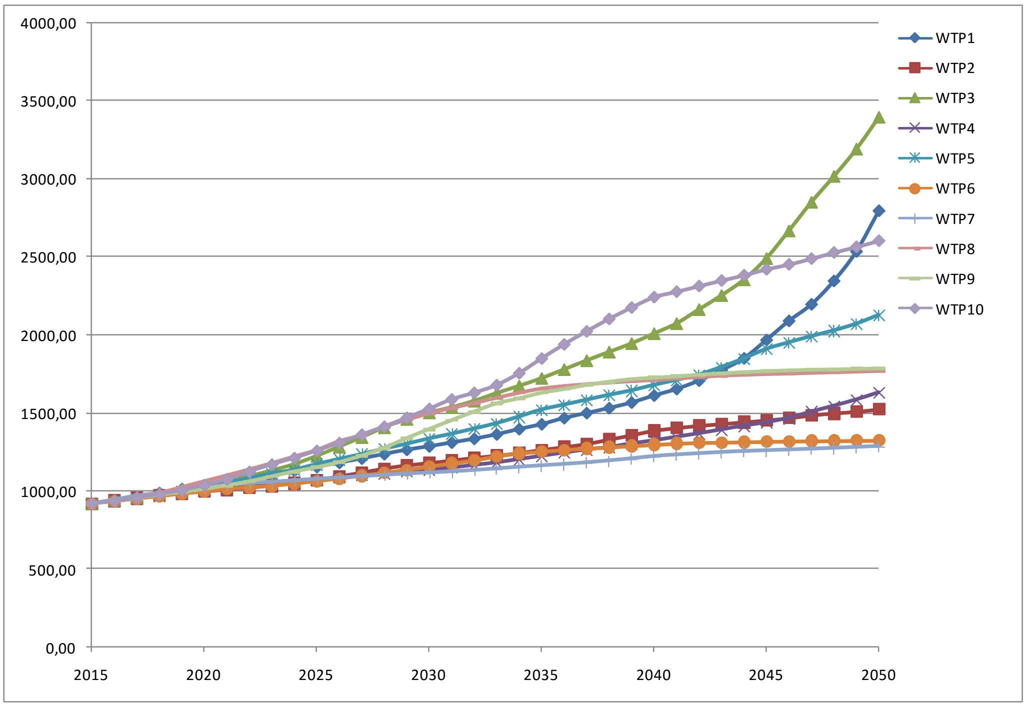



**Fig. 9.** Ten random walk-based future WTP values for the period 2015-2050, for Scenario A

The mean PV of WTP for the Scenario A, which is derived by the 100 repetitions of the random walk model, is around 26,190 Euros (2015). The minimum and maximum PV of WTP estimates are 20,500 and 54,800 Euros (2015) respectively.

In order to quantify the variability due to the range of existing WTP estimates and the uncertainty related to $\alpha_{tot}$ model parameters, the Monte Carlo simulation approach was used running a model for 1,000 times. Following the Maximum Entropy approach, the triangular probability distribution was adopted for all the critical parameters, i.e.

$WTP_{2015}$, the income elasticity of WTP (ω) and the WTP elasticity of demand (λ). More specifically, the likeliest, minimum and maximum values used are as follows:

$WTP_{2015}$ (Euros2015 per ha and per year): min = 0.9; mean = 922; max = 9,345

ω: min = 0.1; mean = 0.7; max = 1

λ: min = -2.3; mean = -1.0; max = -0.5

The results of the Monte Carlo simulation are presented in Tables 13 and 14.

**Table 13.** Monte Carlo simulation summary statistics for the PV of WTP for Scenario A

| Statistics | Forecast value |
|---|---|
| Mean | 93948.03 |
| Median | 85847.00 |
| Standard Deviation | 56159.17 |
| Minimum | 2436.69 |
| Maximum | 278182.65 |
| Mean Std. Error | 1775.91 |

**Table 14.** Monte Carlo simulation percentiles for the PV of WTP for Scenario A

| Percentiles | Forecast values |
|---|---|
| 100% | 2436.69 |
| 90% | 26827.60 |
| 80% | 41609.54 |
| 70% | 54966.25 |
| 60% | 69839.85 |
| 50% | 85842.17 |
| 40% | 101453.39 |
| 30% | 122912.24 |
| 20% | 144693.68 |
| 10% | 173686.92 |
| 0% | 278182.65 |

Additionally, a sensitivity analysis was carried out in order to measure the influence that the input parameters have on the output (i.e. the PV of WTP). To this end, both spider and tornado charts were constructed (Fig. 10 and 11).

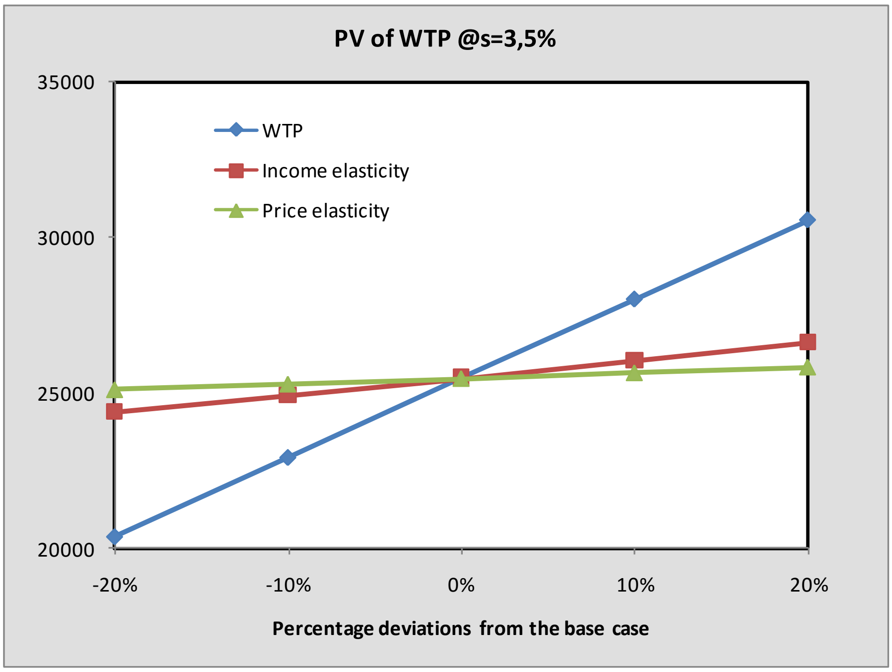



**Fig. 10.** Spider diagram of key input parameters for Scenario A

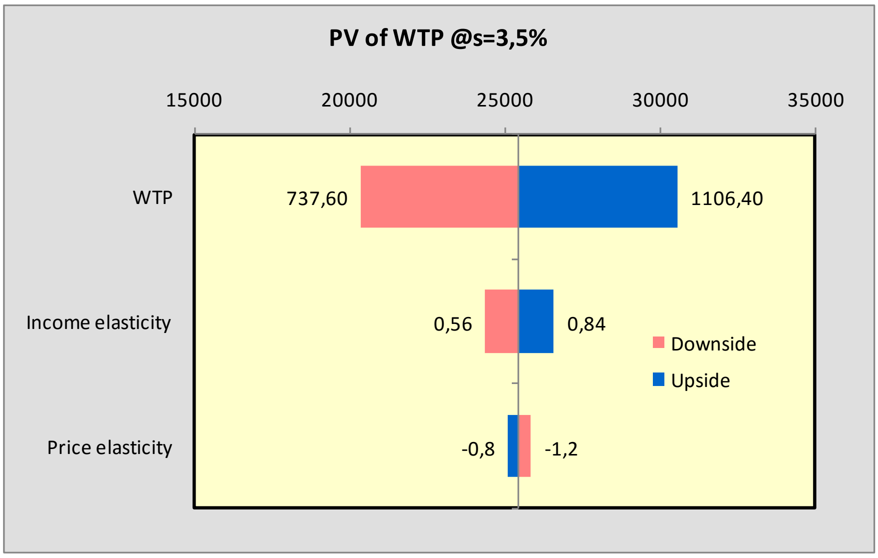



**Fig. 11.** Tornado chart of key input parameters for Scenario A

According to the sensitivity analysis results, the most critical parameter is the $WTP_{2015}$ value, followed by the income elasticity of WTP (ω).

## 6.4.2 Scenario B: Green preferences

According to Scenario B, preferences become greener during the coming decades (i.e. 2015 – 2050). Thus, WTP values are influenced by the growth of income, the decrease in the area of primary forests and the green preferences of the individuals, which are captured in the model by $\alpha_{inc}$, $\alpha_{sc}$ and $\alpha_{pr}$. The values used in the base-case estimates are: $WTP_{2015}$ = 922€; $\alpha_{inc}$ = 1.40% (which is estimated assuming an average growth rate of 2% per year and income elasticity of 0.7); $\alpha_{sc}$ =0.46% (which is estimated assuming an average depletion rate of 0.46% per year and 'environmental' elasticity of WTP of -1); $\alpha_{pr}$ = 1.98% (this value is estimated assuming ratio $WTP_{2050}/WTP_{2015}$ equal to 2); and s =3.5%.

Similarly to the process followed in the case of the Scenario A, 100 repetitions were made to estimate the mean PV, keeping the values of WTP, $\alpha_{inc}$, $\alpha_{sc}$ and $\alpha_{pr}$ constant (Figures 12 and 13 illustrate ten random walks conducted during the modelling process). Thus, the results are only affected by the random component of the model, which is estimated by $\sigma W(t)$.

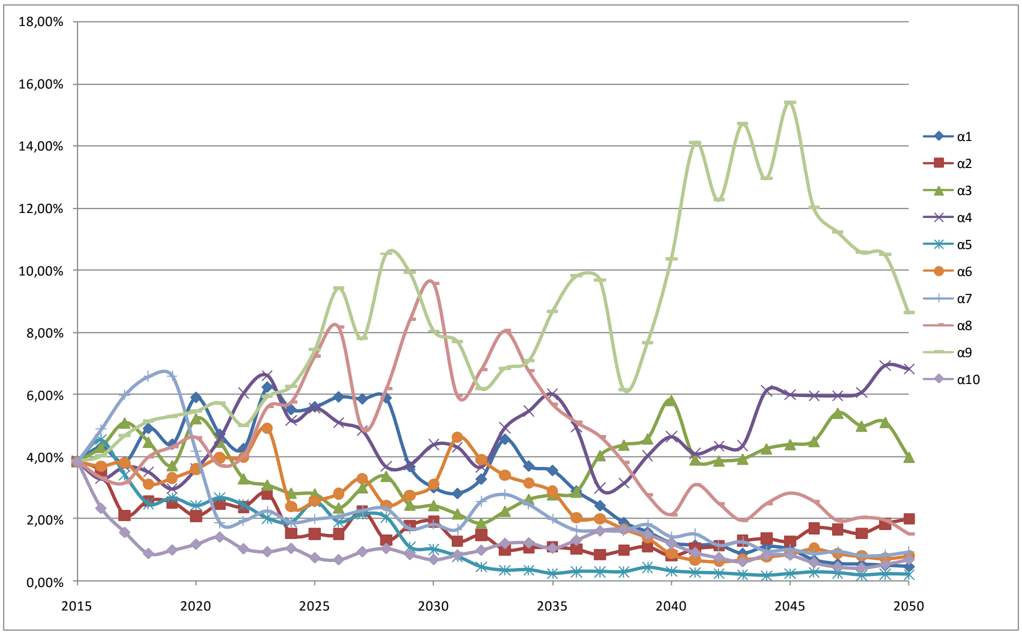



**Fig. 12.** Random walks of $\alpha_{tot}$ for the period 2015-2050, for Scenario B

The mean PV of WTP for the Scenario B, derived by the 100 repetitions of the random walk model, is around 41,830 Euros (2015) and the minimum and maximum estimates are 22,700 and 230,300 Euros(2015) respectively.

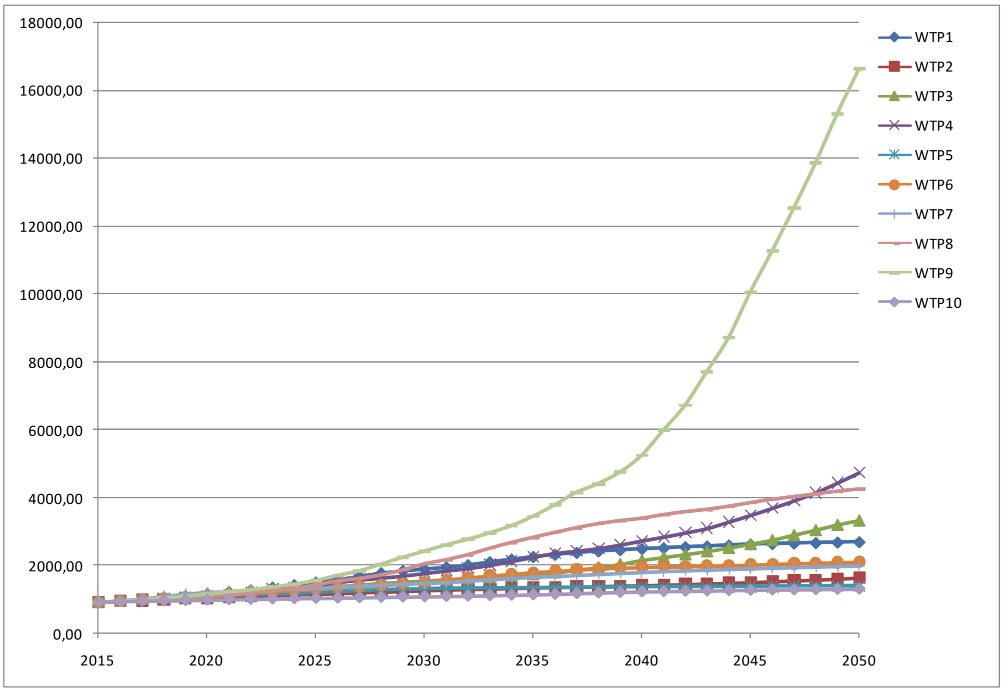



**Fig. 13.** Ten random walk-based future WTP values for the period 2015-2050, for Scenario B

In order to quantify the variability due to the range of exiting WTP estimates and the uncertainty related to the model parameters the Monte Carlo simulation approach was used with 1,000 repetitions. The triangular probability distribution was adopted, according to the Maximum Entropy approach, for the critical parameters, i.e. $WTP_{2015}$, the income elasticity of WTP (ω), the WTP elasticity of demand (λ), and the $WTP_{2050}/WTP_{2015}$ ratio. More specifically, the likeliest, minimum and maximum values used are, as follows:

- $WTP_{2015}$ (Euros2015 per ha and per year): min = 0.9; mean = 922; max = 9,345
- ω: min = 0.1; mean = 0.7; max = 1
- λ: min = -2.3; mean = -1.0; max = -0.5
- $WTP_{2050}/WTP_{2015}$: min = 1.5; mean = 2.0; max = 3.0

The results of the Monte Carlo simulation are presented in Tables 15 and 16.

**Table 15.** Monte Carlo simulation summary statistics for the PV of WTP for Scenario B

| **Statistics** | **Forecast value** |
|---|---|
| Mean | 131073.36 |
| Median | 116015.39 |
| Standard Deviation | 81477.02 |
| Minimum | 5152.00 |
| Maximum | 392361.37 |
| Mean Std. Error | 2576.53 |

**Table 16.** Monte Carlo simulation percentiles for the PV of WTP for Scenario B

| Percentiles | Forecast values |
|---|---|
| 100% | 5152.00 |
| 90% | 34275.61 |
| 80% | 54289.35 |
| 70% | 73028.16 |
| 60% | 93271.72 |
| 50% | 115941.77 |
| 40% | 142829.85 |
| 30% | 170345.59 |
| 20% | 205077.65 |
| 10% | 250554.00 |
| 0% | 392361.37 |

Moreover, a sensitivity analysis was carried out in order to measure the influence that the input parameters have on the output (i.e. the PV of WTP). To this end, both spider and tornado charts were constructed (Fig. 14 and 15).

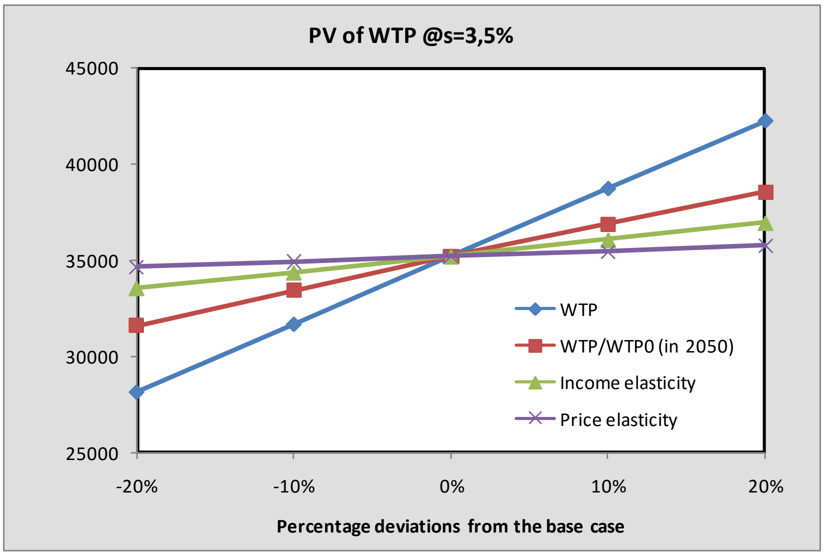



**Fig. 14.** Spider diagram of key input parameters for Scenario B

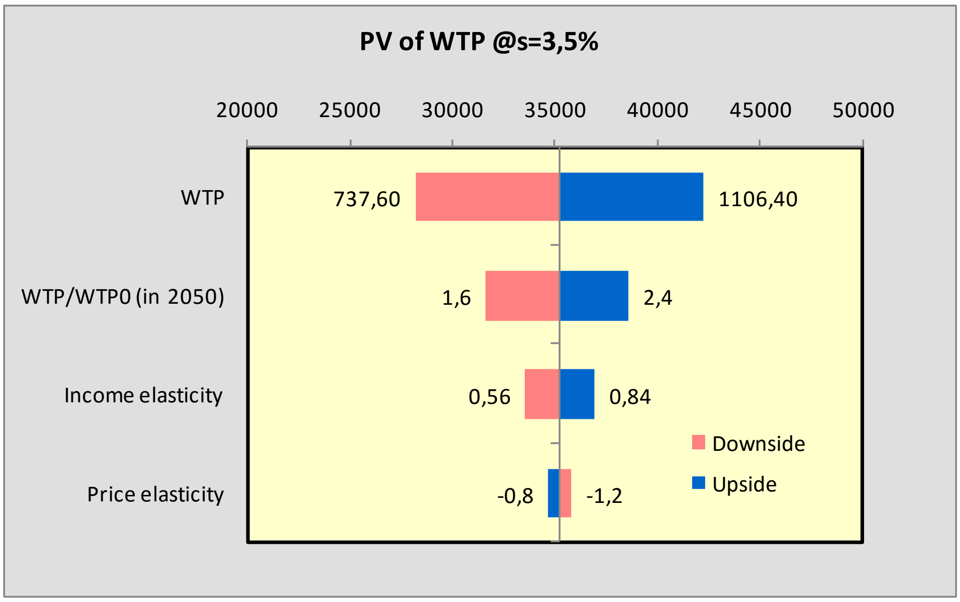



**Fig. 15.** Tornado chart of key input parameters for Scenario B

According to the sensitivity analysis results, the most critical parameter is the $WTP_{2015}$ value, followed by the $WTP_{2050}/WTP_{2015}$ ratio that determines the depletion of the environmental resource, i.e. the primary forests, and the income elasticity of WTP (ω).

### 6.4.3 Scenario C: Materialistic preferences

According to Scenario C, society becomes more materialistic during the coming decades (i.e. 2015 – 2050). Hence, WTP values are influenced by the growth of income, the decrease in the area of primary forests, and the materialistic preferences of the individuals, which are captured in the model by $\alpha_{inc}$, $\alpha_{sc}$ and $\alpha_{pr}$. The values used in the base-case estimates are: $WTP_{2015}$ = 922 €; $\alpha_{inc}$ = 1.40% (which is estimated assuming an average growth rate of 2% per year and income elasticity of 0.7); $\alpha_{sc}$ =0.46% (which is estimated assuming an average depletion rate of 0.46% per year and 'environmental' elasticity of WTP of -1); $\alpha_{pr}$ = -1.98% (this value is estimated assuming ratio $WTP_{2050}/WTP_{2015}$ equal to 0.5); and s =3.5%.

Similarly to the process followed in the case of Scenarios A and B, 100 repetitions were run to estimate the mean PV, keeping the values of WTP, $\alpha_{inc}$, $\alpha_{sc}$ and $\alpha_{pr}$ constant (Figures 16 and 17 illustrate ten random walks conducted during the modelling process). Consequently, the results are only affected by the random component of the model, which is estimated by σW(t). The mean PV of WTP for the Scenario C, according to the 100 repetitions of the random walk model, is approximately 19,050 Euros (2015) and the minimum and maximum estimates are 18,500 and 19,300 Euros (2015) respectively.

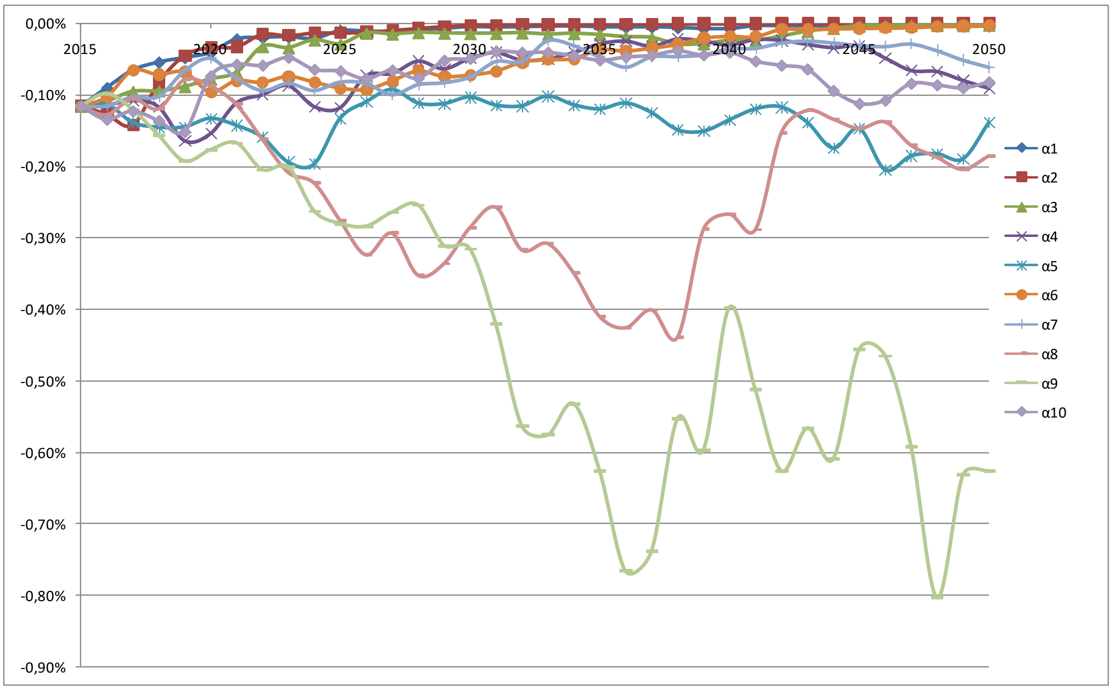



**Fig. 16.** Random walks of $\alpha_{tot}$ for the period 2015-2050, for Scenario C

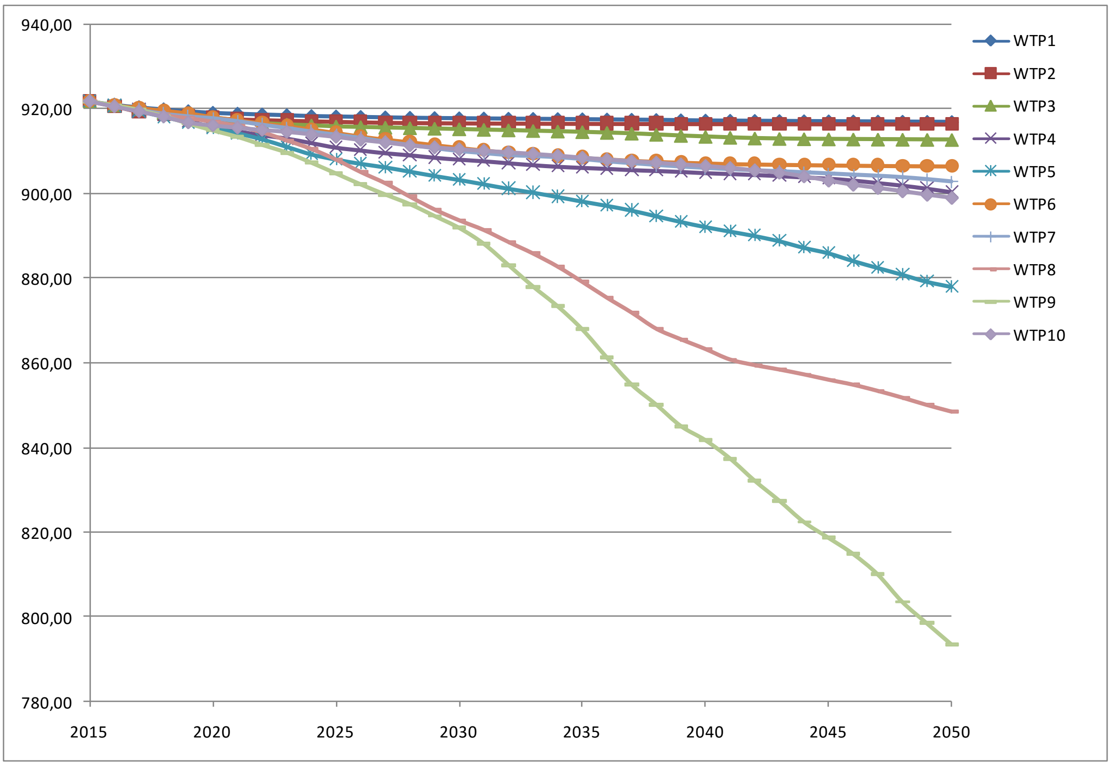



**Fig. 17.** Ten random walk-based future WTP values for the period 2015-2050, for Scenario C

In order to quantify the variability due to the range of existing WTP estimates and the uncertainty related to $\alpha_{tot}$ model parameters, the Monte Carlo simulation approach was used with 1,000 repetitions. Following the Maximum Entropy approach, triangular probability distributions were adopted for the critical parameters, i.e. $WTP_{2015}$, the income elasticity of WTP (ω), the WTP elasticity of demand (λ), and the $WTP_{2050}/WTP_{2015}$ ratio. More specifically, the likeliest, minimum and maximum values used are, as follows:

$WTP_{2015}$ (Euros2015 per ha and per year): min = 0.9; mean = 922; max = 9,345

ω: min = 0.1; mean = 0.7; max = 1

λ: min = -2.3; mean = -1.0; max = -0.5

$WTP_{2050}/WTP_{2015}$: min = 0.33; mean = 0.5; max = 0.67

The results of the Monte Carlo simulation are presented in Tables 15 and 16.

**Table 17.** Monte Carlo simulation summary statistics for the PV of WTP for Scenario C

| Statistics | Forecast value |
|---|---|
| Mean | 70304.31 |
| Median | 64117.55 |
| Standard Deviation | 43818.93 |
| Minimum | 206.79 |
| Maximum | 209339.84 |
| Mean Std. Error | 1385.68 |

**Table 18.** Monte Carlo simulation percantiles for the PV of WTP for Scenario C

| Percentiles | Forecast values |
|---|---|
| 100% | 206.79 |
| 90% | 17823.65 |
| 80% | 28831.33 |
| 70% | 39480.41 |
| 60% | 52116.19 |
| 50% | 64116.55 |
| 40% | 76170.61 |
| 30% | 89940.73 |
| 20% | 107206.58 |
| 10% | 134893.30 |
| 0% | 209339.84 |

Furthermore, a sensitivity analysis was carried out in order to measure the influence that the input parameters have on the output (i.e. the PV of WTP). To this end, both spider and tornado charts were constructed (Fig. 14 and 15).

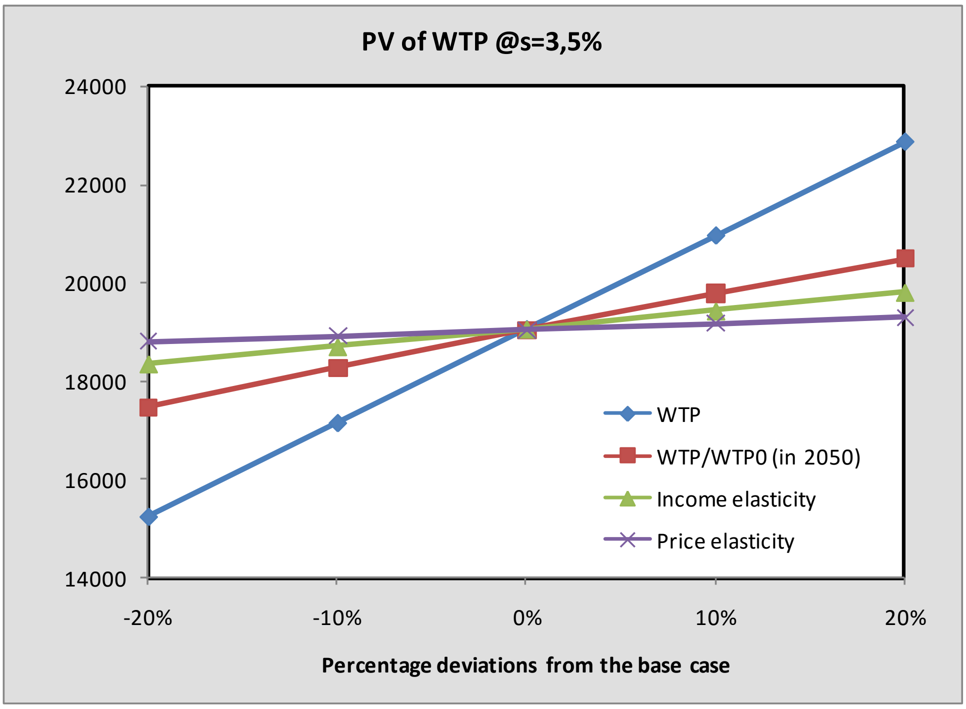



**Fig. 18.** Spider diagram of key input parameters for Scenario C

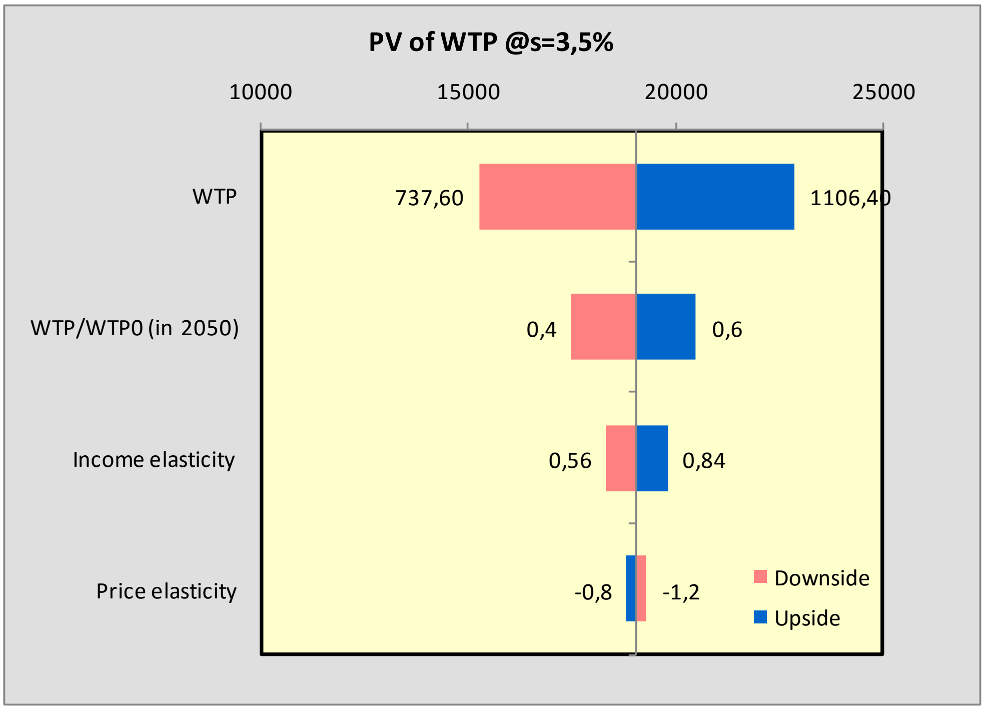



**Fig. 19.** Tornado chart of key input parameters for Scenario C

Similarly to the sensitivity analysis results for Scenario B, the most critical parameter is the $WTP_{2015}$ value, followed by the $WTP_{2050}/WTP_{2015}$ ratio that determines the depletion of the environmental resource, i.e. the primary forests, and the income elasticity of WTP (ω).

## 6.5 Discussion of results

Based on the analysis presented in Sections 6.3 and 6.4, the following remarks follow:

**A. PV of WTP: Deterministic vs. probabilistic assessments**

In all scenarios studied, i.e. with and without considering the effect of total growth rate $\alpha_{tot}$, the probabilistic assessment results in higher estimates than the deterministic analysis. To wit, the mean PV of WTP derived by the probabilistic simulation when the growth rate $\alpha_{tot}$ is ignored is around 72,200 Euros (2015), while the deterministic estimate is 19,360 Euros (2015). The mean PV of WTP for the Scenario A, which is derived by the 100 repetitions of the random walk model, is around 26,190 Euros (2015). The probabilistic mean PV of WTP for the same Scenario is around 94,000 Euros (2015), i.e. around 2.6 times higher. Similar results are drawn for Scenarios B and C.

The disparities between the deterministic estimates and the probabilistic simulations are attributed primarily to the wide range of the $WTP_{2015}$ value, i.e. between 0.9 and 9,345 $Euros_{2015}$ per ha and per year with mean = 922 $Euros_{2015}$ per ha and per year.

**B. PV of WTP: with and without considering the total growth rate $\alpha_{tot}$**

The effect of total growth rate $\alpha_{tot}$ on the estimated PV of WTP is significant, even when preferences are assumed to remain constant. This is evident when comparing the results presented in Section 6.3 with those of Section 6.4.1 (i.e. the Scenario A). More specifically, the mean (probabilistic) PV of WTP for Scenario A (i.e. 94,000 Euros2015) is 30% higher than the one estimated ignoring the growth rate $\alpha_{tot}$ (i.e. around 72,200 Euros2015).

The effect of total growth rate $\alpha_{tot}$ is even more apparent when changes in the preferences of individuals are involved in the stochastic model. For instance, in the case of green preferences (i.e. Scenario B), the mean (probabilistic) PV of WTP is around 132,000 Euros(2015), that is almost twice the PV estimated ignoring the growth rate $\alpha_{tot}$.

**C. PV of WTP: with and without changing preferences**

The comparison of the estimates for the three Scenarios A, B and C reveals the importance of considering the effect of evolving preferences, especially in long-term analyses. In fact, all the other parameters being equal, the estimated PV of WTP is quite different for constant preferences, green preferences and materialistic preferences. More specifically, the PV of WTP for the scenario of constant preferences (i.e. Scenario A) is about 35% higher than the estimated PV for the case of materialistic preferences (i.e. Scenario C), and about 30% lower than the PV estimated for the case of green preferences (i.e. Scenario B).

More importantly, the comparison between the estimates of Scenarios B and C highlights the significance of the assumptions adopted regarding the evolution of preferences in the next decades. To wit, the PV estimated for Scenario B, which corresponds to greening of preferences, is around 88.5% higher than the corresponding value of Scenario C. This finding is worrisome, considering that future preferences are unknown since complex and interlinked socioeconomic and behavioural factors are involved, which are also changeable. Therefore, potential behavioural patterns should be considered in the analyses, at least for sensitivity purposes.

# 7 Conclusions and a look ahead

The present report has investigated the issue of future preferences and changing values as a central object of inquiry referring to the estimation of climate change damages. The analytical encounter with the problem of future preferences is central to the economics of adaptation assessment since the economic rationale of investing in adaptation projects strongly hinges on the estimation of avoided future damages. Our random walk model allows the analyst to visualize future paths of preference and value evolution and by doing so brings future values of damaged assets realistically to the fore.

The reliability of the model crucially depends on the reliability of input data referring to the elasticities of demand and income as well as projected growth rate of world economies. A step forward therefore is the embedding of our model into reliable Shared Socioeconomic Pathways (SSPs); this would add to the completeness and reliability of parameter estimation and integrate the model into the wider discussion of socio-economic pathways for adaptation assessment.

# References


Agthe, D. and Billings R. (1980). Dynamic models of residential water demand. Water Resource Research 16, pp. 476–480.

Arbués, F., García-Valiñas, M. Á., &Martínez-Espiñeira, R. (2003). Estimation of residential water demand: A state-of-the-art review. Journal of Socio-Economics, 32(1), pp. 81-102.

Ayong Le Kama, A., & Schubert, K. (2004). Growth, Environment and Uncertain Future Preferences. Environmental and Resource Economics, 28(1), pp. 31-53.

Ayres, R. U., Bergh, J. C. J. M. V. D., &Gowdy, J. (1998). Viewpoint: weak versus strong sustainability. Tinbergen Inst. Discuss. Pap. Tinbergen Institute.

Bacchetta, P., & Van Wincoop, E. (2007). Random walk expectations and the forward discount puzzle. American Economic Review, 97(2), pp. 346-350.

Barbier, E.B., Baumgärtner, S., Chopra,K., Costello, C., Duraiappah, A., Hassan, R., Kinzig, A., Lehman, M., Pascual, U., Polasky, S., Perrings, C. (2009). The Valuation of Ecosystem Services. Chapter 18. In: Naeem S., D. Bunker, A. Hector, M. Loreau and C. Perrings (eds.), Biodiversity, Ecosystem Functioning, and Human Wellbeing: An Ecological and Economic Perspective. Oxford University Press, Oxford, UK, pp. 248–262.

Bergstrom, T.C. and Goodman, R.P. (1973). Private demands for public goods. American Economic Review 63 (3), pp. 280–296.

Bindé J. (2004). The Future of Values. 21st-Century Talks. Berghahn Editions, N. York and Oxford

Borcherding, T.E. and Deacon, R.T. (1972). Demand for services of non-federal governments. American Economic Review 62 (5), pp. 891–901.

Borges, M. R. (2011). Random walk tests for the Lisbon stock market. Applied Economics, 43(5), pp. 631-639.

Bosetti, V., Tavoni, M., De Cian, E., Sgobbi, A. (2009). The 2008 WITCH Model: New Model Features and Baseline. FEEM Working Paper 2009.085.

Brouwer, R., Brander, R., Kuik, O., Papyrakis, E. and Bateman, I. (2013). A synthesis of approaches to assess and value ecosystem services in the EU in the context of TEEB. Report to the European Commission. http://ec.europa.eu/environment/nature/biodiversity/economics/pdf/EU%20Valuation.pdf

Cairns, R. D. (2014). The green paradox of the economics of exhaustible resources. Energy Policy, 65, pp. 78-85.

Costa, D. L., & Kahn, M. E. (2002). Changes in the Value of Life, 1940–1980. NBER Working Paper No. 9396. http://www.nber.org/papers/w9396

Cynthia Lin, C. Y., & Wagner, G. (2007). Steady-state growth in a Hotelling model of resource extraction. Journal of Environmental Economics and Management, 54(1), pp. 68-83.

Dalhuisen, J., Florax, R., Groot, H., & Nijkamp, P. (2003). Price and income elasticities of residential water demand: a meta-analysis. Land Economics, 79, pp. 292-308.

Daly, H. (1995). On Wilfred Beckerman's Critique of Sustainable Development. Environnemental Values, 4(1), pp. 49-55.

Davis, D.W. (2000). Individual level examination of postmaterialism in the U.S.: Political tolerance, racial attitudes, environmentalism, and participatory norms. Political Research Quarterly, 53, pp. 455–475.

DEFRA (2007). An introductory guide to valuing ecosystem services. London, UK, 65 pp.

Dolan P. and A. Tsuchiya (2005). Health priorities and public preferences: the relative importance of past health experience and future health prospects. Journal of Health Economics 24: 703–714

Doremus H. (2003). Shaping the Future: The Dialectic of Law and Environmental Values, 27 Environs: Environmental. Law and Policy Journal 233, available at: http://scholarship.law.berkeley.edu/facpubs/615

Duracz, A. (2006). Derivation of Probability Distributions for Risk Assessment, Master´s Thesis in Computer Science at Stockholm University, Sweden.

Edwards, N., Grepin, H., Haurie, A., Viguier, L. (2005). Linking Climate and Economic Dynamics. In: Coupling of Climate and Economic Dynamics: Essays on Integrated Assessment, Eds. Alain Haurie and Laurent Viguier, Amsterdam: Springer.

Espey, M., Espey, J., & Shaw, W. D. (1997). Price elasticity of residential demand for water: A meta-analysis. Water Resources Research, 33(6), pp. 1369 – 1374.

Fama, E. F. (1965). Random Walks in Stock Market Prices. Financial Analysts Journal, 21(5), pp. 55-59.

Fisher, A. C. and J. V. Krutilla (1974). Valuing Long-Run Ecological Consequences and Irreversibilities, Journal of Environmental Economics and Management 1, pp. 96–108.

Fisher, A.C. and Krutilla, J.V. (1975). Resource conservation, environmental preservation, and the rate of discount. Quarterly Journal of Economics, 89(3), pp. 358-370.

Flores, N.E. and Carson, R.T. (1997). The Relationship between the Income Elasticities of Demand and Willingness to Pay. Journal of Environmental Economics and Management, 33, pp. 287-295.

Freeman, M., III. (2003). The measurement of environmental and resource values: Theory and methods, 2nd ed., Resources for the Future.

Gay, C. and Estrada, F. (2010). Objective probabilities about future climate are a matter of opinion, Climatic Change, 99, pp. 27-46.

Goksen, F., Adaman, F., Zenginobuz, U. (2002). On environmental concern, willingness to pay, and postmaterialist values Evidence from Istanbul. Environment and Behavior, 34, 5, pp. 616-633

Goulder, L.H., Kennedy, D. (1997). Valuing ecosystem: philosophical bases and empirical methods. In: G.C. Daily (Ed.), Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington D.C., pp. 23–48.

Gravelle H. and D. Smith (2000).Discounting for Health Effects in Cost Benefit and Cost Effectiveness Analysis. Centre for Health Economics Technical Paper Series 20. University of York.

Grijalva, T. C., Lusk, J. L., & Shaw, W. D. (2013). Discounting the Distant Future: An Experimental Investigation. Environmental and Resource Economics, 59(1), pp. 39-63.

Groom, B., Hepburn, C., Koundouri, P., & Pearce, D. (2005). Declining discount rates: The long and the short of it. Environmental and Resource Economics, 32(4), pp. 445-493.

Groot, R. de, Brander, L., Ploeg, S. van der, Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services, 1(1), pp. 50-61.

H.M. Treasury (2003). The Green Book: Appraisal and Evaluation in Central Government. London.

Hanemann, M. (1991). Willingness to pay and willingness to accept: How much can they differ? The American Economic Review, 81(3), pp. 635-647.

Harr, M.E. (1987). Reliability-Based Design in Civil Engineering, McGraw-Hill, New York.

Hewitt, J., and Hanemann, M. (1995). A Discrete-continuous Choice Approach to Residential Water Demand under Block Rate Pricing. Land Economics 71(2), pp. 173–192.

Hoel, M. and Sterner, T. (2007). Discounting and relative prices. Climatic Change, 84(3-4), pp. 265-280

Hökby, S. and Söderqvist, T. (2003). Elasticities of Demand and Willingness to Pay for Environmental Services in Sweden. Environmental and Resource Economics, 26(1980), pp. 361-383.

Hope, C.W. (2006). The marginal impact of CO2 from PAGE2002: An integrated assessment model incorporating the IPCC's five reasons for concern. Integrated Assessment Journal, 6(1), pp. 19–56.

Horowitz, J.K. (2002). Preferences in the future. Environmental and Resource Economics, 21(3), pp. 241-259.

Horowitz, J.K. and McConnell, K.E. (2003). Willingness to accept, willingness to pay and the income effect. Journal of Economic Behavior and Organization, 51(4), pp. 537-545.

Hotelling, H. (1931). The Economics of Exhaustible Resources, Journal of Political Economy, 39(2), pp. 139– 175.

Hultman, M., Kazeminia, A., Ghasemi, V. (2015). Intention to visit and willingness to pay premium for ecotourism: The impact of attitude, materialism, and motivation. Journal of Business Research, 68, pp. 1854–1861.

Inglehart, R. (1981). Post-materialism in an environment of insecurity. American Political Science Review, 75, pp. 880–900.

Jacobsen, J. B., & Hanley, N. (2009). Are there income effects on global willingness to pay for biodiversity conservation? Environmental and Resource Economics, 43, pp. 137-160.

Jacobsen, J. B., Lundhede, T. H. & Thorsen, B. J. (2013). The effects of current income and expected change in future income on stated preferences for environmental improvements. Journal of Forest Economics, 19(2), pp. 206-219.

Jaynes, E.T. (1957). Information theory and statistical mechanics, Physical Review, 106(4), pp. 620–630.

Khan, H. (2009). Willingness to pay and demand elasticities for two national parks: Empirical evidence from two surveys in Pakistan. Environment, Development and Sustainability, 11(2), pp. 293-305.

Khan, H. (2012). Estimating elasticities of demand and willingness to pay for clean drinking water: empirical evidence from a household level survey in northern Pakistan, Water and Environment Journal, 28, pp. 145–152.

Kidd, Q. and Lee, A.R. (1997). Postmaterialist values and the environment: A critique and reappraisal. Social Science Quarterly, 78, pp. 1-15.

Kilbourne, W. and Pickett, G. (2008). How materialism affects environmental beliefs, concern, and environmentally responsible behavior. Journal of Business Research, 61, pp. 885–893.

Kodukula, P. and Papudesu, P. (2006). Project Valuation Using Real Options: A Practitioner's Guide, J. Ross Publishing: N. Andrews Way, Florida.

Kontogianni A., Tourkolias, C., Machleras, A. and Skourtos, M. (2012). Service providing units, existence values and the valuation of endangered species: A methodological test. Ecological Economics 79, pp. 97–104.

Kontogianni, A., Tourkolias, C.H., Damigos, D. and Skourtos, M. (2013). Assessing sea level rise costs and adaptation benefits under uncertainty in Greece. Environmental Science & Policy, 37, pp. 61-78.

Kopp R.E. and B. K. Mignone (2012). The U.S. Government's Social Cost of Carbon Estimates After Their First Two Years; Pathways for Improvement. Economics: The Open-Access, Open-Assessment EJournal, Vol. 6, 2012-15. http://dx.doi.org/10.5018/economics-ejournal.ja.2012-15

Kristrom, B. and Riera, P. (1996). Is the income elasticity of environmental improvements less than one? Environmental & Resource Economics, 7(1), pp. 45-55.

Krysiak, F.C. and Krysiak, D. (2006). Sustainability with Uncertain Future Preferences. Environmental & Resource Economics, 33, pp. 511–531.

Ludwig, L. and Neumann, J. (2012). Updating Income Elasticity Estimates in EPA's BenMAP Air Pollution Benefits Estimation System, Memorandum March 30 , 2012. http://www3.epa.gov/ttnecas1/regdata/Benefits/IncomeElasticityUpdate_Recommendationswithappendices.pdf

Manne, I. and Richels, R. (2004). MERGE: An Integrated Assessment Model for Global Climate Change.

Manomaivibool P. and S. Vassanadumrongdee (2012). Buying back household waste electrical and electronic equipment: Assessing Thailand's proposed policy in light of past disposal behavior and future preferences. Resources, Conservation and Recycling 68: 117–125

Martini, C. and Tiezzi, S. (2013). Is the environment a luxury? An empirical investigation using revealed preferences and household production. Resource and Energy Economics, 37, pp. 147-167.

McFadden, D. (1994). Contingent valuation and social choice, American Journal of Agricultural Economics, 76, pp. 689-708.

Meyer, M. A. and Booker, J. M. (2001). Eliciting and Analyzing Expert Judgment: A Practical Guide, ASA-SIAM Series on Statistics and Applied Probability, 459 p.

Minnitt, R.C.A. (2007). Frontiers of usefulness: The economics of exhaustible resources. Journal of the Southern African Institute of Mining and Metallurgy, 107(8), pp. 539-552.

Mishra, S. (2002). Assigning probability distributions to input parameters of performance assessment models, SKB Technical Report TR-02-11.

Mun, J., 2006. Real Options Analysis - Tools and Techniques for Valuing Strategic Investments and Decisions, 2nd ed. John Wiley & Sons, Inc.: Hoboken, NJ.

Myersa D. and E. Gearinb (2001). Current preferences and future demand for denser residential environments. Housing Policy Debate 12: 633-659

Newell, R.G., & Pizer, W.A. (2003). Discounting the distant future: How much do uncertain rates increase valuations? Journal of Environmental Economics and Management, 46(1), pp. 52-71.

Noblet C.L., M. W. Anderson, M. F. Teisl (2015). Thinking past, thinking future: An empirical test of the effects of retrospective assessment on future preferences. Ecological Economics 114 (2015) 180–187

Nordhaus, W., and Boyer, J. (2000). Warming the World: Economic Models of Global Warming. Cambridge, MA: MIT Press.

Nordhaus, W., and Sztorc, P. (2013). DICE 2013R: Introduction and User's Manual 2nd edition. http://www.econ.yale.edu/~nordhaus/homepage/documents/DICE_Manual_103113r2.pdf

Nunes, P. and van den Bergh, J. (2001). Economic Valuation of Biodiversity: Sense or Nonsense? Ecological Economics 39(2), pp. 203–222.

OECD (2012a). Medium and long-term scenarios for global growth and imbalances, Ch. 4. OECD Economic Outlook, Volume 2012/1. http://www.oecd.org/berlin/50405107.pdf

OECD (2012b). Environmental outlook to 2050: The consequences of Inaction. Key Findings on Biodiversity.

OECD (2015). Inflation (CPI). https://data.oecd.org/price/inflation-cpi.htm

Ortiz, R.A. and Markandya, A. (2009). Integrated Impact Assessment Models of Climate Change with an Emphasis on Damage Functions: a Literature Review. BC3 Working Paper Series.

Parry, M., Arnell, N., McMichael, T., Nicholls, R., Martens, P., Kovats, S., Livermore, M., Rosenzweig, C., Iglesias, A., and Fischer, G. (2001). Millions at risk: Defining critical climate change threats and targets. Global Environmental Change-Human And Policy Dimensions 11(3), pp. 181–83.

Pattanayak, S., Wing, J., Depro, M., Van Houtven, G., De Civita, P., Stieb, D. & Hubbell, B. (2002). International health benefits transfer application tool: the use of PPP and inflation indices. Final report, prepared for Economic Analysis and Evaluation Division, Office of Policy Coordination and Economic Analysis Policy and Planning Directorate, Healthy Environments and Consumer Safety Branch, Health Canada.

Pearce, D., Markandya, A. & Barbier, E.B. (1997). Blueprint for a Green Economy, Earthscan Publications, London.

Peck, S.C. and Teisberg, T.J. (1992). CETA: A model for carbon emissions trajectory assessment. Energy Journal, 13, 1, pp. 55–77.

Pesando, J. (1979). On the Random Walk Characteristics of Short- and Long-Term Interest Rates In an Efficient Market. Journal of Money, Credit and Banking, 11(4), pp. 457-466.

Pezzey, J.C.V. (2006). Reconsidering reconsidered: Why sustainable discounting need not be inconsistent over time, in Pannell, D. &Schilizzi, S. (eds), 'Discounting and Discount Rates in Theory and Practice', Edward Elgar, Cheltenham.

Pint, E.M. (1999). Household Responses to Increased Water Rates during the California Drought. Land Economics 75(2), pp. 246–266.

PricewaterhouseCoopers (2015). The World in 2050 - Will the shift in global economic power continue? https://www.pwc.com/gx/en/issues/the-economy/assets/world-in-2050-february-2015.pdf

Rollins, K., &Lyke, A. (1998). The Case for Diminishing Marginal Existence Values. Journal of Environmental Economics and Management, 36(3), pp. 324-344.

Rosenberger, R.S. & Stanley, T.D. (2010). Publication selection of recreation demand price elasticity: a meta-analysis. https://www.hendrix.edu/uploadedFiles/Departments_and_Programs/Business_and_Economics/AMAES/MAER%202010%20AR%20Rosenberger%20Stanley%20090110.pdf

Scalas, E. (2006). The application of continuous-time random walks in finance and economics. Physica A: Statistical Mechanics and its Applications, 362(2), pp. 225-239.

Scarborough, H. (2010). Decomposing the social discount rate. AARES Conference. http://ageconsearch.umn.edu/bitstream/59156/2/Scarborough,%20Helen.pdf

Scheierling, S. M., Loomis, J. B., & Young, R.A. (2006). Irrigation water demand: A meta-analysis of price elasticities. Water Resources Research, 42(1), pp. 1-9.

Shafer, G. & Vovk, V. (2001). Game Theoretic Probability in Finance. In: Probability and Finance: It's Only a Game! Wiley: New York, pp. 201-236.

Shannon, C.E. (1948). A mathematical theory of communication, The Bell System Technical Journal, 27, pp. 379–423, 623–656.

Sieg, H., Smith, V. K., Banzhaf, H.S and Walsh, R. (2000). Estimating the general equilibrium benefits of large policy changes: the Clean Air Act revisited. National Bureau of Economic Research, Working Paper 7744. Cambridge MA: NBER.

Skourtos M., A. Kontogianni, P. Harrison (2010). Reviewing the dynamics of economic values and preferences for ecosystem goods and services. Biodiversity and Conservation, 19: 2855-2872

Smith, G. (2002). Tests of the Random Walk Hypothesis for London Gold Prices. Applied Economics Letters, 9(10), pp. 671-674.

Stanton, E.A., Ackerman, F., Kartha, S. (2008). Inside Integrated Assessment Models: four issues in climate economics. Stockholm Environment Institute Working Paper, WP-US-0801.

Sterner, T. & Persson, U. M. (2008). An even sterner review: Introducing relative prices into the discounting debate. Review of Environmental Economics and Policy, 2(1), pp. 61-76.

Summers L. and R. Zeckhauser (2008). Policymaking for posterity. Journal of Risk and Uncertainty 37: 115-140

TEEB (2010). The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundation, Earthscan, London.

Thomas, J. F., and Syme, G. J. (1988). Estimating residential price elasticity of demand for water: A contingent valuation approach. Water Resources Research, 24(11), pp. 1847-1857.

Thunström L., J. Nordström, J. F. Shogren (2015). Certainty and overconfidence in future preferences for food. Journal of Economic Psychology 51 (2015) 101–113

Tietenberg, T.H. (1996). Environmental and natural resource economics (4th ed.).New York: HarperCollins.

Tol, R. (1999). Spatial and temporal efficiency in climate policy: Applications of FUND. Environmental and Resource Economics, 14, pp. 33–49.

Tol, R. and Frankhauser, S. (1998). On the representation of impact in integrated assessment models of climate change. Environmental Modelling and Assessment, 3, pp. 63-74.

Toth, F.L. (2005). Coupling Climate and Economic Dynamics: recent achievements and unresolved problems., In: Coupling of Climate and Economic Dynamics: Essays on Integrated Assessment, Eds. Alain Haurie and Laurent Viguier, Amsterdam: Springer.

Traeger, C. (2007), Theoretical aspects of long-term evaluation in environmental economics, PhD thesis, University of Heidelberg. http://www.ub.uni-heidelberg.de/archiv/7049

Turner, R.K., Pearce, D. and Bateman I. (1994). Environmental economics: An elementary introduction. Harvester Wheatsheaf, Hertfordshire, U.K.

U.S. Department of Energy (2010). Energy conservation program: Energy conservation standards for small electric motors; Final Rule. Federal Register, 75(45): 10874–10948. URL http://www1.eere.energy.gov/buildings/appliance_standards/ commercial/small_electric_motors.html.

United Nations World Commission on Environment and Development - WCED (1987). Our common future. UN General Assembly Document. Oxford: Oxford University Press. p. 27.

Waldhoff, S., Anthoff, D., Rose, S., Tol, R. (2014). The Marginal Damage Costs of Different Greenhouse Gases: An Application of FUND. Economics Discussion Paper.

Witsenhausen, H.S. (1974). On the Uncertainty of Future Preferences. Annals of Economic and Social Measurement 3, pp. 91–94.

World Bank (2015). PPP conversion factor, GDP (LCU per international $). From: http://data.worldbank.org/indicator/PA.NUS.PPP?page=4 (November 25, 2015)

Worthington, A. C., & Hoffman, M. (2008). An empirical survey of residential water demand modelling. Journal of Economic Surveys, 22(5), pp. 842-871.

Yoo, J., Simonit, S., Kinzig, A.P. and Perrings, C. (2014). Estimating the Price Elasticity of Residential Water Demand: The Case of Phoenix, Arizona. Applied Economic Perspectives and Policy 36(2), pp. 333-350.

Yu, X., Gao, Z., Zeng, Y. (2014). Willingness to pay for the ‘‘Green Food’’ in China. Food Policy, 45, pp. 80–87.

# Appendix

The monetary values used by de Groot et al. (2012) per service for Temperate and Boreal Forests are provided in the following table.

**Table A.1.** List of used monetary values per service for Temperate and Boreal Forests

| ES Service | | Value | Valuation Method | Country | Reference |
|---|---|---|---|---|---|
| 1 | Food | 0 | DMP | Canada | Anielski and Wilson (2005) |
| 1 | Food | 597 | DMP | Mexico | Adger et al. (1994) |
| 2 | Water | 118 | FI / PF | Chile | Nunez et al. (2006) |
| 2 | Water | 333 | RC | China | Xue and Tisdell (2001) |
| 2 | Water | 121 | DMP | Portugal | Cruz and Benedicto (2009) |
| 3 | Raw materials | 59 | DMP | Canada | Anielski and Wilson (2005) |
| 3 | Raw materials | 2 | DMP | Samoa | Mohd-Shahwahid and McNally (2001) |
| 3 | Raw materials | 2 | DMP | Samoa | Mohd-Shahwahid and McNally (2001) |
| 3 | Raw materials | 662 | DMP | Eritrea | Emerton and Asrat (1998) |
| 8 | Climate | 7 | RC | Canada | Anielski and Wilson (2005) |
| 8 | Climate | 23 | DMP | Canada | Anielski and Wilson (2005) |
| 8 | Climate | 186 | DMP | Mexico | Adger et al. (1994) |
| 8 | Climate | 36 | DMP | Mexico | Adger et al. (1994) |
| 8 | Climate | 624 | MC / RC | China | Xue and Tisdell (2001) |
| 8 | Climate | 33 | RC | China | Xue and Tisdell (2001) |
| 11 | Waste | 0 | AC | Mexico | Adger et al. (1994) |
| 11 | Waste | 0 | AC | Mexico | Adger et al. (1994) |
| 11 | Waste | 22 | RC | Portugal | Cruz and Benedicto (2009) |
| 12 | Erosion | 5 | AC | China | Xue and Tisdell (2001) |
| 13 | Soil fertility | 93 | DMP | China | Xue and Tisdell (2001) |
| 15 | BioControl | 469 | RC | Sweden | Hougner et al. (2006) |

| ES Service | | Value | Valuation Method | Country | Reference |
|---|---|---|---|---|---|
| 15 | BioControl | 1 | AC | China | Xue and Tisdell (2001) |
| 17 | Genepool | 3.151 | CV | Finland | Kniivila et al. (2002) |
| 17 | Genepool | 3.573 | CV | USA | Loomis and Ekstrand (1998) |
| 17 | Genepool | 790 | CV | Finland | Siikamäki and Layton (2007) |
| 17 | Genepool | 68 | CV | South Africa | Turpie (2003) |
| 17 | Genepool | 130 | CV | USA | Walsh et al. (1984) |
| 17 | Genepool | 212 | CV | Finland | Siikamäki and Layton (2007) |
| 17 | Genepool | 487 | CV | Finland | Siikamäki and Layton (2007) |
| 17 | Genepool | 51 | CV | USA | Walsh et al. (1984) |
| 17 | Genepool | 67 | CV | USA | Walsh et al. (1984) |
| 17 | Genepool | 88 | CV | USA | Walsh et al. (1984) |
| 19 | Recreation | 1 | CV | Samoa | Mohd-Shahwahid and McNally (2001) |
| 19 | Recreation | 1 | TC | Italy | Bellu and Cistulli (1997) |
| 19 | Recreation | 9 | CV | Sweden | Bostedt and Mattsson (2006) |
| 19 | Recreation | 88 | CV | Denmark | Dubgaard (1998) |
| 19 | Recreation | 10.027 | Other | Finland | Kniivila et al. (2002) |
| 19 | Recreation | 169 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 429 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 9.398 | CV | Netherlands | Van der Heide (2005) |
| 19 | Recreation | 12 | TC | Portugal | Cruz and Benedicto (2009) |
| 19 | Recreation | 406 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 1.240 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 7 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 14 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 36 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 115 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 116 | CV | Ireland | Scarpa et al. (2000) |

| ES Service | | Value | Valuation Method | Country | Reference |
|---|---|---|---|---|---|
| 19 | Recreation | 138 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 1.014 | CV | Ireland | Scarpa et al. (2000) |
| 19 | Recreation | 73 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 125 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 191 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 240 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 266 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 305 | CV | United Kingdom | Scarpa et al. (2000) |
| 19 | Recreation | 306 | CV | United Kingdom | Scarpa et al. (2000) |
| 22 | Cognitive | 1 | TC | Portugal | Cruz and Benedicto (2009) |

Source: de Groot et al. (2012), Supplementary data associated with this article

# Sub-task 2: Report on treatment of future learning and quasi-option values

Basque Centre for Climate Change – BC3, Bilbao

# Executive Summary

This report was produced for Work Package 2 of the FP7 ECONADAPT project, which carries out research in the economics of adaptation to man-made climate change. WP2 of the project on *the micro-economics of adaptation* reviews and develops methods to better provide empirical data for undertaking the economic assessment of adaptation, focussing on a number of data parameters that are currently poorly characterised for the adaptation context. This report specifically presents methods to consider future values for adaptation assessments.

This report addresses two approaches to incorporate uncertainty into decision-making in relation to climate change adaptation. The first approach is the Real Options Analysis (ROA); the second is based on risk measures and the concept of acceptable risk.

# 1 Introduction to the problem

There is limited evidence and a great deal of uncertainty about many impacts of climate change, especially precipitation (Jiménez Cisneros et al., 2014). At the same time socioeconomic factors pay a major part in determining the consequences of climatic and other extreme events. During the 20th century flood-damages have risen as a result of a greater exposure and vulnerability of assets and people, and the contribution of socio-economic factors to flood risk has been estimated to be equal or even greater than that of climate change alone (Jiménez Cisneros et al., 2014; Kovats et al., 2014).

In this context, decisions to invest on adaptation need to deal with a major issue: uncertainty (Hallegatte, 2009). The different kinds of uncertainty can be classified according to a matrix developed by Refsgaard et al. (2007), which identifies the three dimensions of uncertainty in an environmental modelling framework: first, the “nature of uncertainty”, i.e. whether this is due to incomplete knowledge or is the result of natural variability; second, the “type of uncertainty”, e.g. statistical uncertainty or that related to scenarios; third, the “source of uncertainty”, that can be related to the context under study, input data, etc. Under climate change, it is important to note that cumulative uncertainty is most likely to occur, as every step of each dimension will add uncertainty to the following one (Markandya, 2014). In other words, in order to assess an investment for adaptation, we need to acknowledge that there will be uncertainty related to the context (the study area), the climate modelling, the impact modelling, the socio-economic scenarios, etc. Some authors consider that developing approaches that account for uncertainty is one of the main priorities in the field of economics of adaptation to climate change (Hunt and Watkiss, 2010).

As theory moves to practice, adaptation has been acknowledged to be dynamic, as preferences may vary with time as new or improved climate information is available or technologies arise or evolve. This being so, robust approaches that consider flexibility and the time dimension can be very valuable to support decision-making under uncertainty (Chambwera et al., 2014).

From the methodological perspective, a robust analysis can be defined based on three components (Markandya, 2014). The first consists of assessing the robustness of measures. Measures are defined as robust when they are effective in a wide range of future scenarios. Typically, low- and no-regret measures provide robustness in situations of uncertainty about the future. However, some of these measures that are able to cope with a wide variety of scenarios can be too costly; others, such as early warning systems, while being cheaper, will not be enough to cope with some extreme situations, for example, the 500 years return period floods, and will likely not prevent all damage in the event of any flood.

The second component relates to flexibility in decision making. In this case, low- and no-regret options could be decided at the short term, waiting for more and better information or technologies to implement the costliest policies.

Finally, the third component analyses the adaptability of options in response to future information or needs. For example, building a dyke with foundations strong enough for a 2 m-high wall, that could be built in the future.

In this note we consider two ways in which uncertainty can be incorporated into decision-making in relation to adaptation to climate change. One is the use of Real Options Analysis (ROA). The other is to use the concept of acceptable risk and develop a method for applying it in this context. Section 2 describes how ROA may be used and provides an example of an application to flood risk reduction in the Basque Country. Section 3 outlines the use of acceptable risk and how it might be applied to the evaluation of flood protection in a global context for major cities.

## 2 Real Options Analysis

Real options analysis has evolved from the financial economics literature and is intended to deal with future uncertainties of a project's implementation (Zeng and Zhang, 2011). The concept is relatively easy to understand: when an investment decision is made, the entity undertaking it can obtain a right that can be used to buy or sell a physical asset or investment plan linked to the investment decision in the future (Myers, 1977).

In the context of adaptation economics, it can be said that "ROA quantifies the investment risk with uncertain future outcomes" (Watkiss et al., 2015: 407). "This includes the flexibility over the timing of the capital investment, but also the flexibility to adjust the investment as it progresses over time, i.e. allowing a project to adapt, expand or scale-back in response to unfolding events. The approach can therefore assess whether it is better to invest now or to wait – or whether it is better to invest in options that offer greater flexibility in the future." (Watkiss and Hunt, 2013).

This investment analysis tool has been widely used in the energy sector (e.g. Abadie et al., 2014) but it has gained much interest recently in the framework of adaptation economics as it "aligns with the concepts of iterative adaptive (risk) management, providing a means to undertake economic appraisal of future option values the value of information and learning, and the value of flexibility, under conditions of uncertainty. It can therefore justify options (or decisions) that would not be taken forward under a conventional economic analysis" (Watkiss and Hunt, 2013).

There are, however, relatively few applications for adaptation alternatives or investment projects using ROA. One exception is Kontogianni et al. (2014) where the alternatives to protect the Greek coast from sea level rise are analysed. The authors conclude that the analysis "through recognizing the uncertainty and keeping all the options open till uncertainty is resolved, provides an adaptation strategy that may be beneficial […] for the society". Another interesting example can be found in Jeuland and Whittington (2013) with an application to water resource planning in Ethiopia for the construction of several large dams and operating strategy accounting for uncertainties due to climate change. And a third example is Woodward et al. (2011) for flood risk management in the Thames Estuary. The authors conclude that "the results obtained demonstrate the potential for substantial cost savings under future uncertainties when Real Options are used instead of more traditional, precautionary approaches".

Against this background, a study a rigorous application of ROA has been undertaken for a public investment in infrastructure planned to reduce flood-risk in the city of Bilbao (Basque Country, Spain), which involves opening an pre-existing canal that will turn the current peninsula of Zorrotzaurre into an island in the Bilbao Estuary. In order to do so, a stochastic model has been developed, which contains two risk variables: the frequency of extreme flood events and the stochastic growth rate of damage, as a function of climate change and socio-economic development. Based on that an ROA is carried out to obtain the wait and investment regions for different discount rates and volatility values.

## 2.1 Application of ROA to the City of Bilbao

Bilbao is the main city and economic engine of the Basque Country (Spain). The city has a population of nearly 350,000 but the metropolitan area that extends from Bilbao towards the sea along the estuary gathers 850,000 people, 40 percent of the population of the region (Eustat, 2014). The Bilbao estuary was once the most extensive estuarine area along the Cantabrian coast (Hazera, 1968), but during the last two centuries it has been dramatically reshaped by conversion of land to industrial and urban occupation.

The Basque Country is an area with high-risk due to natural flood hazard. This flood risk is the result of natural hazard (high precipitation, strong slopes and steep valleys), resulting in a high vulnerability, with most of its low-lying areas densely urbanised (Ibisate et al., 2000). The region has suffered several flooding episodes in its recent history, which have caused significant damages. The most catastrophic flooding event occurred in August 1983, during which 37 people died and material damages rose up to current 1,206 M€, probably the costliest flood event in Spain in economic terms (Olcina et al., 2016). The city of Bilbao was one of the most affected.

Like other old industrial cities, the urban development in Bilbao has been shaped by the requirements of the manufacturing industry accompanied by a fast growing population (Rodríguez et al., 2001). Most of this urban expansion during the mid-20th century occurred in flood prone areas along the estuary, which increased the vulnerability of the city. After the dramatic floods in 1983, several infrastructure measures were implemented (Fernández Gómez, 1993), but the risk still remains. For example, in May 2008 the water reached its highest levels in 20 years and a new severe flood did not happen because this time high peak levels met a low tide (Diputación Foral de Bizkaia, 2008).

In 2012 a new important urban development was approved in an old industrial site located on the peninsula of Zorrotzaurre, a flood-prone area in the Bilbao Estuary. As a response to the strong concern of the Basque Water Agency (URA) in relation to the development of a new urban district in an area subject to severe risk of flooding, the option of opening of the Deusto canal, which would turn Zorrotzaurre into an island significantly reducing the risk of flooding upstream, was put forward for consideration. This measure was finally approved and the works are currently underway.

Because of its history, and more so for its infrastructural implications, its flood prevention capacity and its engaged investments, the opening of the Deusto canal is considered a major adaptation measure for Bilbao.

## 2.2 The costs and benefits of the infrastructure

The opening of the Deusto canal that will turn Zorrotzaurre into an island is estimated to significantly reduce flood risk not only on the island, but also in several other areas of Bilbao by increasing the drainage capacity of the estuary. The intervention consists of a 75 m wide opening, which would reduce the water level by an average of 0.87 m for the 500-years return period. In some areas the difference of the water level with or without the intervention could be as high as 1.43 m (SAITEC, 2007). The cost of this measure is estimated at 12.1 M EUR and it will be financed entirely by the Bilbao City Council. Figure 1 shows the land areas as at present and with the canal.

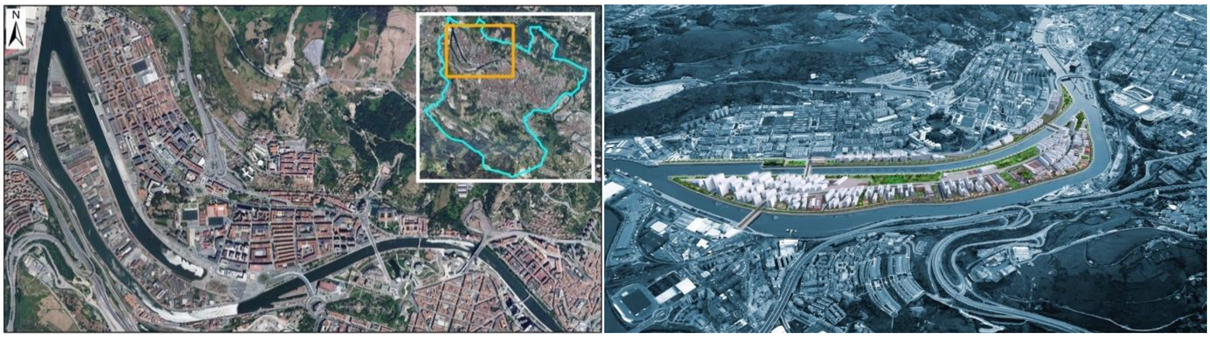


**Figure 1.** Aerial view of Zorrotzaurre and its location in the municipality of Bilbao (left) and a simulation of the area after the opening of the Deusto canal, including new developments in its northern part.

A previous study commissioned by the Bilbao City Council estimated the economic benefits of the opening of the Deusto canal in terms of avoided damages (Osés Eraso et al., 2012). The study assessed floods with 10, 100 and 500 year return periods, whose main features were known: flood-extension, depth and water speed. This information was combined with socio-economic data about those elements exposed to the risk of flooding. As in most cities, in Bilbao the main elements at risk are houses, shops, businesses, historic buildings and citizens.

Baseline damages, before the opening of the canal, were taken from a study by the Basque Government (2007). The same methodology was used to develop the damage function in both studies but, in order to define the new adaptation scenario, several new variables were incorporated in the analysis:

- The opening-width of the canal of 50 metres[1].
- The new water level varies from 1.07 m in the baseline[2] to 0.70 m after the opening, so an average level of 0.885 m was considered. This new level was considered to be equal in every section.

[1] The final width of the canal is 70 m therefore the benefits of adaptation are expected to be even higher than those estimated by Osés Eraso et al. (2012).

[2] According to the report commissioned by URA to SAITEC (2007).

- No new data on water speed was available, so it was assumed to be the same as in the baseline.

Five categories of damages were used to estimate the new costs of flooding after the opening of the Deusto canal. The first category accounts for damages to residential property, which can be classified into direct costs (to property, furniture or other appliances, including cleaning costs) and indirect damages (relocation). Direct costs estimates were transferred to Bilbao based on a study developed in the UK (Penning-Rowsell et al., 2006). These costs depend on water depth, the type of housing, the age of the affected buildings, the social class and the duration of flooding. Relocation costs were based on another study in the UK (DETR, 1999). The second category of costs includes damages to non-residential property, which accounts for damage to (non-residential) buildings, machinery or stored items and indirect damage due to a possible temporary cessation of activity. These estimates were also based on Penning-Rowsell et al. (2006).

The third type of damage is related to impacts on cultural heritage, which were based on a study from Taylor (2006) that used a contingent valuation method to obtain the willingness to pay to avoid the risk of flooding in two buildings of heritage interest in Lewes (UK).The results were transferred to Bilbao. The fourth type of damages refer to flood impacts on human health which may result from the event itself (risk to life, hypothermia and injuries during or immediately after) and from the subsequent activities related to the event (stress, post-traumatic anxiety…). Estimates for health damages were based on several studies from DEFRA (2003, 2004, 2006). This category includes foregone benefits related to the willingness to pay for increasing the level of protection, and is closely related to anxiety resulting from previously experienced events. A fifth category of damages was also accounted for, which temporary disruption of transportation, increasing the number of emergencies and so -called second -round effects following the approach by Penning-Rowsell et al. (2006).

The results obtained by Osés Eraso et al. (2012) show a significant reduction of damages in the adaptation scenario. Floods of 10 year return period would not cause any damage, while costs decrease by 67.4% for 100 years return period floods. For 500-year return period floods, damages are reduced by 30.7%. The results are presented in Table 1.

### 2.3 Flood-damages and the benefits of adaptation in Zorrotzaurre: a stochastic approach

The first step in the ROA is to estimate expected damages at different points in time. To do this a stochastic function was developed, assuming that the intensity of the extreme events does not change.

The expected damage $E(D)$ in an interval $dt$ can be expressed as:

$$E(D) = E(D_1) + E(D_2) + E(D_3) = E(d_1 \times d^1 q + d_2 \times d^2 q + d_3 \times d^3 q) = \quad (1)$$

$$d_1 \lambda_1 dt + d_2 \lambda_2 dt + d_3 \lambda_3 dt \quad (2)$$

Where D stands for damage for floods of different return periods (i). Where the independent Poisson process $d^i q$ has a value of 1 with probability $\lambda_i \times dt$ and 0 otherwise. $\lambda_i$ is the return period of flooding (i.e. the frequency). In this example, $\lambda_1$would represent the 10-year return-period floods; $\lambda_2$the 100-year return period and $\lambda_3$ the 500-year return period.

The expected damage between the initial time $\tau_1 = 0$ and the final time $\tau_2$ can be presented as:

$$E(D_i^{0,\tau_2}) = \int_0^{\tau_2} d_i \lambda_i e^{-\rho t} dt = \frac{d_i \lambda_i}{\rho}[1 - e^{-\rho \tau_2}] \qquad (3)$$

where $\rho$ is the discount rate with risk and $d_i$ is the corresponding value from Table 3 that indicates the damage if there is a flood event of type $i$.

In the long run, we can consider that time tends to infinite ($\tau_2 \to \infty$), and thus Equation 3 can be simplified as follows:

$$E(D_i^{0,\infty}) = \int_0^{\infty} d_i \lambda_i e^{-\rho t} dt = \frac{d_i \lambda_i}{\rho} \qquad (4)$$

**Table 1.** Avoided damages (benefits) resulting from the opening of the Deusto canal. Baseline flood damages are taken from Basque Government (2007) and damages after building the canal were estimated by Osés Eraso et al. (2012). Data is shown in millions of euros per event.

| Category of damage | APF = 1/10 | | | | | | APF = 1/100 | | | | | | APF = 1/500 | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Baseline | | Opening | | Benefits | | Baseline | | Opening | | Benefits | | Baseline | | Opening | | Benefits | |
| | Low | High | Low | High | Low | High | Low | High | Low | High | Low | High | Low | High | Low | High | Low | High |
| **Direct property damage** | | | | | | | | | | | | | | | | | | |
| Residential property | 4.67 | 5.72 | 0 | 0 | **4.67** | **5.72** | 164.83 | 197.59 | 61.36 | 73.49 | **103.47** | **124.10** | 235.15 | 276.45 | 192.05 | 228.34 | **43.10** | **48.11** |
| Non-residential property | 0 | 0 | 0 | 0 | **0** | **0** | 24.67 | 25.95 | 0 | 0 | **24.67** | **25.95** | 101.03 | 106.26 | 41.81 | 43.98 | **59.22** | **62.28** |
| Cultural heritage | 0 | 0 | 0 | 0 | **0** | **0** | 0.20 | 2.01 | 0.20 | 2.01 | **0.00** | **0.00** | 1.02 | 10.13 | 1.02 | 10.13 | **0.00** | **0.00** |
| **Other effects** | | | | | | | | | | | | | | | | | | |
| Temporary accommodation | 0.04 | 0.04 | 0 | 0 | **0.04** | **0.04** | 1.07 | 1.07 | 0.40 | 0.40 | **0.67** | **0.67** | 1.68 | 1.68 | 1.35 | 1.35 | **0.33** | **0.33** |
| Additional power use | 0.26 | 0.26 | 0 | 0 | **0.26** | **0.26** | 7.56 | 7.56 | 2.77 | 2.77 | **4.79** | **4.79** | 8.68 | 8.68 | 8.13 | 8.13 | **0.55** | **0.55** |
| Health (anxiety) | 0.02 | 0.02 | 0 | 0 | **0.02** | **0.02** | 0.61 | 0.61 | 0.22 | 0.22 | **0.39** | **0.39** | 0.67 | 0.67 | 0.65 | 0.65 | **0.02** | **0.02** |
| Health (injuries and fatalities) | 0.03 | 0.16 | 0 | 0 | **0.03** | **0.16** | 13.22 | 26.89 | 6.76 | 13.18 | **6.46** | **13.71** | 46.38 | 80.14 | 28.24 | 50.32 | **18.14** | **29.82** |
| Emergency services | 0.50 | 0.61 | 0 | 0 | **0.50** | **0.61** | 20.28 | 23.78 | 6.57 | 7.86 | **13.71** | **15.92** | 35.97 | 40.39 | 25.02 | 28.91 | **10.95** | **11.48** |
| Forgone profit | 0 | 0 | 0 | 0 | **0** | **0** | 8.30 | 8.30 | 0 | 0 | **8.30** | **8.30** | 12.19 | 12.19 | 8.30 | 8.30 | **3.89** | **3.89** |
| Rail disruption | 0 | 0 | 0 | 0 | **0** | **0** | 0.21 | 0.21 | 0.21 | 0.21 | **0.00** | **0.00** | 0.86 | 0.86 | 0.86 | 0.86 | **0.00** | **0.00** |
| Secondary effects | 0.01 | 0.01 | 0 | 0 | **0.01** | **0.01** | 0.38 | 0.45 | 0.12 | 0.15 | **0.26** | **0.30** | 0.67 | 0.79 | 0.47 | 0.56 | **0.20** | **0.23** |
| **TOTAL** | 5.53 | 6.82 | | | **5.53** | **6.82** | 241.33 | 294.42 | 78.61 | 100.29 | **162.72** | **194.13** | 444.30 | 538.24 | 307.90 | 381.53 | **136.40** | **156.71** |

As we are considering floods with return periods of 10, 100 and 500 years, there is an expected damage $E(D_i^{0,\tau_2})$ for each frequency $\lambda_i$. When there are three types of events the present value of the total expected damage for the interval $[0,\tau_2]$ is:

$$E(D^{0,\tau_2}) = E(D_1^{0,\tau_2}) + E(D_2^{0,\tau_2}) + E(D_3^{0,\tau_2}) \quad (5)$$

## 2.4 The effect of climate change and socio-economic development on expected damage

Future damages could increase due to the effect of climate change (more frequent and/or more intense flood events) but also due to socio-economic development, that is, the value of the assets at risk is expected to be higher in the future. In addition, the new urban development of the Zorrotzaurre district may imply a significantly higher number of assets that could be potentially affected by flooding.

If climate and socio-economic effects are factored in, damage could increase at a rate of $\mu_C$ due to climate effects and $\mu_S$ due to socio-economic effects, so that the total increase can be defined as the sum of both effects: $\mu = \mu_C + \mu_S$. As the growth rate and the discount rate have opposite signs, the expected damage will depend on the difference between the two. In other words, if event $i$ takes place at time $t$ the expected damage in the interval $[0,\tau_2]$ is as follows:

$$E(D_i^{0,\tau_2}) = \int_0^{\tau_2} d_i \lambda_i e^{(\mu-\rho)t} dt = \frac{d_i \lambda_i}{\rho - \mu}[1 - e^{-(\rho-\mu)\tau_2}] \quad (6)$$

As previously done, when $\tau_2 \to \infty$ then:

$$E(D_i^{0,\infty}) = \int_0^{\infty} d_i \lambda_i e^{(\mu-\rho)t} dt = \frac{d_i \lambda_i}{\rho - \mu} \quad (7)$$

Note that Equation (7) would only make sense if $\mu < \rho$. However, in principle there is nothing to prevent the sum of the climate and socio-economic effects from being greater than or equal to the discount rate. In such a case we should consider Equation (6) for a finite time intervals.

In summary, the stochastic damage function defined by Equation (6) enables the calculation of expected flooding costs for any given time, depending on the difference between the discount rate and the sum of the increase of damages due to climate change and the economic growth. Using the data from Table 1 as an input, we can stochastically measure the expected flood damages of different return periods, but similarly, we can also estimate the benefits (in terms of avoided impacts) of the opening of the Deusto canal. The results of the stochastic modelling are shown in Table 2.

**Table 2.** Expected reduction of damage that results from opening the Deusto canal, for different values of $\rho - \mu$ and different time periods ($\tau$, in years).

| $\rho - \mu$ | Damage base case (1) | | | Island Case (2) | | | Damage Reduction (1)-(2) | | |
|---|---|---|---|---|---|---|---|---|---|
| | $\tau_2$=50 | $\tau_2$=85 | $\tau_2$=100 | $\tau_2$=50 | $\tau_2$=85 | $\tau_2$=100 | $\tau_2$=50 | $\tau_2$=85 | $\tau_2$=100 |
| -0.02 | 367.75 | 957.52 | 1,367.40 | 136.09 | 354.35 | 506.03 | 231.66 | 603.18 | 861.37 |
| -0.01 | 277.68 | 573.43 | 735.50 | 102.76 | 212.21 | 272.18 | 174.92 | 361.22 | 463.32 |
| 0 | 214.02 | 363.84 | 428.04 | 79.20 | 134.64 | 158.40 | 134.82 | 229.19 | 269.64 |
| 0.01 | 168.42 | 245.09 | 270.58 | 62.33 | 90.70 | 100.13 | 106.10 | 154.39 | 170.44 |
| 0.02 | 135.29 | 174.92 | 185.06 | 50.07 | 64.73 | 68.48 | 85.22 | 110.19 | 116.57 |
| 0.03 | 110.84 | 131.54 | 135.58 | 41.02 | 48.68 | 50.17 | 69.83 | 82.86 | 85.41 |
| 0.04 | 92.53 | 103.44 | 105.05 | 34.24 | 38.28 | 38.88 | 58.29 | 65.16 | 66.18 |
| 0.045 | 85.10 | 93.05 | 94.06 | 31.49 | 34.43 | 34.81 | 53.60 | 58.61 | 59.25 |
| 0.05 | 78.58 | 84.39 | 85.03 | 29.08 | 31.23 | 31.47 | 49.50 | 53.16 | 53.56 |
| 0.06 | 67.79 | 70.91 | 71.16 | 25.09 | 26.24 | 26.34 | 42.70 | 44.67 | 44.83 |
| 0.065 | 63.30 | 65.59 | 65.75 | 23.42 | 24.27 | 24.33 | 39.87 | 41.32 | 41.42 |
| 0.07 | 59.30 | 60.99 | 61.09 | 21.95 | 22.57 | 22.61 | 37.36 | 38.42 | 38.48 |

Observe that a wide range of options for $\rho - \mu$ have been presented in Table 2. The social discount rate for water investments in Spain reached 8% (Groom, 2014), so we use this value of $\rho$ for case study of Zorrotzaurre. In relation to $\mu$, we need to consider climate and socio-economic effects separately. The expected economic growth for the European Union[3] in terms of GDP per capita based on purchasing power parity (PPP) ranges between 1.6% in 2050 and 1.1% by the end of the century, based on estimates by different organisms[4], so this could be a reasonable range for $\mu_S$ in the Basque Country as well.

As mentioned in the Introduction, damage is expected to depend to a greater extent on socio-economic development rather than the effect of climate change. We thus assume that $\mu_c$ could vary between 0% (no effect) up to 1.6% (equal to the highest economic growth rate). Therefore, $\mu$ could range between 1.1% and 3.2%. The result is that the probable range for $\rho - \mu$ in Zorrotzaurre would be 4.5-6.5%.

---

[3] Members prior to 2004, excluding countries that joined the EU since then, as they may have a different growth path.

[4] Based on estimates of SSP2 from IIASA, OECD and PIK. Full data available at: https://secure.iiasa.ac.at/web-apps/ene/SspDb/dsd?Action=htmlpage&page=about

The reduction of damages due to the adaptation measure of opening the Deusto canal can also be observed in Figure 2, including the probable range of $\rho - \mu$ for Zorrotzaurre.

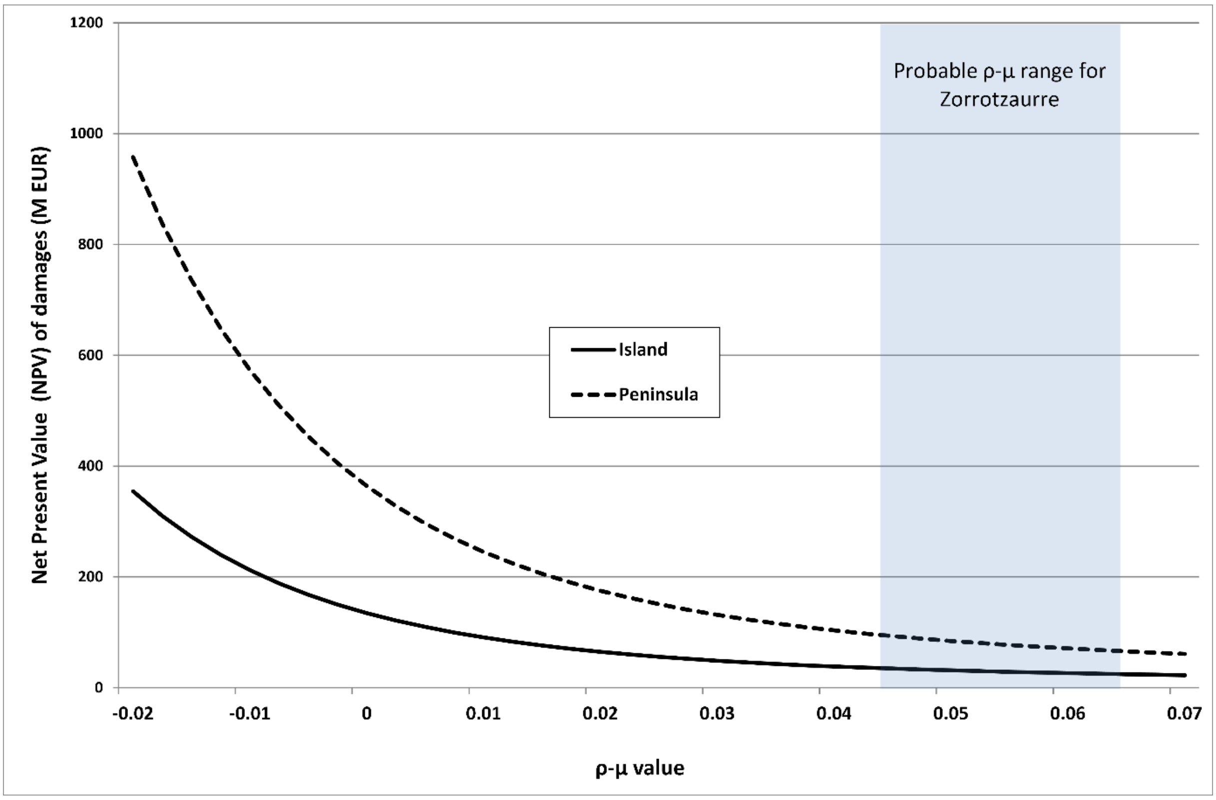



**Figure 2.** Net present value of the damage for the baseline (Zorrotzaurre as a peninsula) and the adaptation scenario, once the opening of the canal is finished and the Zorrotzaurre district is an island, as a function of $\rho - \mu$. Time is 100 years ($\tau_2 = 100$).

## 2.5 Including Uncertainty into the Value of Damages

There are two main risk measurements that can be used for this purpose: Value-at-Risk (*VaR*) and Expected Shortfall (ES). The first one is the most a standard measurement and well recognised by international financial regulatory bodies. The *VaR(α)* at the confidence level *α* is the value at which the probability of obtaining higher values is *1-α*. In our case study, the VaR of damage resulting from river flooding in the Bilbao Estuary expresses the losses that could occur with a given confidence level *α* of 95%, for a time interval of 100 years.

The second risk measure is the Expected Shortfall (ES), which in our case represents the expected damage when VaR is exceeded. ES is, therefore, a better measure of risk for low probability but high damage events and a more robust indicator to assess risk (Rockafellar and Uryasev, 2002). Both measures of risk have been estimated for the Bilbao case study.

The opening of the canal is expected to reduce not only the expected damage but also the level of risk, that is, the damages that would occur in the worst 5% of the cases A risk assessment follows, performed using MonteCarlo simulations[5], for the baseline where Zorrotzaurre is a peninsula and the adaptation scenario, once the Deusto canal is opened. The time interval considered is 85 years (period 2015-2100, $\tau_2 = 85$) and the rate is 6.5% ($\rho - \mu = 0.065$). One million MonteCarlo simulations were run, each with 50 steps per annum $\Delta t = 1/50$. Results are presented in Figure 3, which shows that both the expected damage and the risks decrease significantly due to the opening of the canal. For example, in the island case of the million simulations performed, we found 360,926 where damage is zero, which is equivalent to say that there is a 36.1% probability of there being no damage due to 100 and 500 year flood events.

The MonteCarlo method is needed to estimate the distribution of damages probabilities for different events so that we can calculate the probability of exceeding different levels of damages. However, it is important to check that the method yields results consistent with those generated by the underlying stochastic process for flood occurences. This can reasonably be represented as a Poisson process. The Poisson distribution indicates the likelihood of a certain number of events happening within a given interval:

$$f(k, \Lambda_i) = \frac{e^{-\Lambda_i} \Lambda_i^k}{k!} \tag{8}$$

where $k$ is the number of events and $\Lambda_i$ is the expected value in an interval of 85 years ($T = \tau_2 - \tau_1$). Following this rule, we can estimate, for example, the possibility of not having any flood event in 85 years (period 2015-2100) due to the opening of the canal:

- For 100-year floods ($i = 2; T = 85$): $\Lambda_2 = T \times \lambda_2 = 0.85$
- For 500-year floods ($i = 3; T = 85$): $\Lambda_3 = T \times \lambda_3 = 0.17$

Note that $i = 1$ , which represents the 10-year return-period events, have not been included because it is expected that these events would produce no damage after the opening of the canal.

Once $\Lambda_i$ is known, we can estimate the probability of zero events of both kinds of flood events:

- For 100-year floods ($i = 2; T = 85$): $f(0, \Lambda_2) = e^{-\Lambda 2} = e^{-0.85} = 0.4274$
- For 500-year floods ($i = 3; T = 85$): $f(0, \Lambda_3) = e^{-\Lambda 3} = e^{-0.17} = 0.8437$

5 Monte Carlo methods are tools often use to deal with complex probability distributions (Mosegaard and Sambridge, 2002). Here we simulate damages in time. One million paths are simulated, each divided in periods of 1/50 years. Then the net present value of each period is estimated in order to calculate damages for each patch. All the paths are equally probable, thus we can obtain a distribution of frequencies and estimate risk measures.

The probability of there being no damage of any kind by 2100, within 85 years after the opening up of the canal with the types of event being independent is:

$$f(0, \Lambda_2) \times f(0, \Lambda_3) = 0.3606$$

The distribution of probabilities in 85 years time interval estimated analytically following Equation (8) is presented in Table 3. This analytic result is very close to that obtained vía the MonteCarlo simulation, where a probability of 0.3609 was obtained (360,926 cases per million, see Figure 4B). This results shows the high accuracy of MonteCarlo simulation, which we use in order to be able to estimate the risk and the values of *VaR(α)* and ES.

Based on MonteCarlo simulations, two measures of risk have been estimated, as a function of *ρ-μ*: the VaR(95%) and ES(95%). Table 4 shows the expected damage value and the risk measures in the island and peninsula cases as a function of *ρ-μ*:

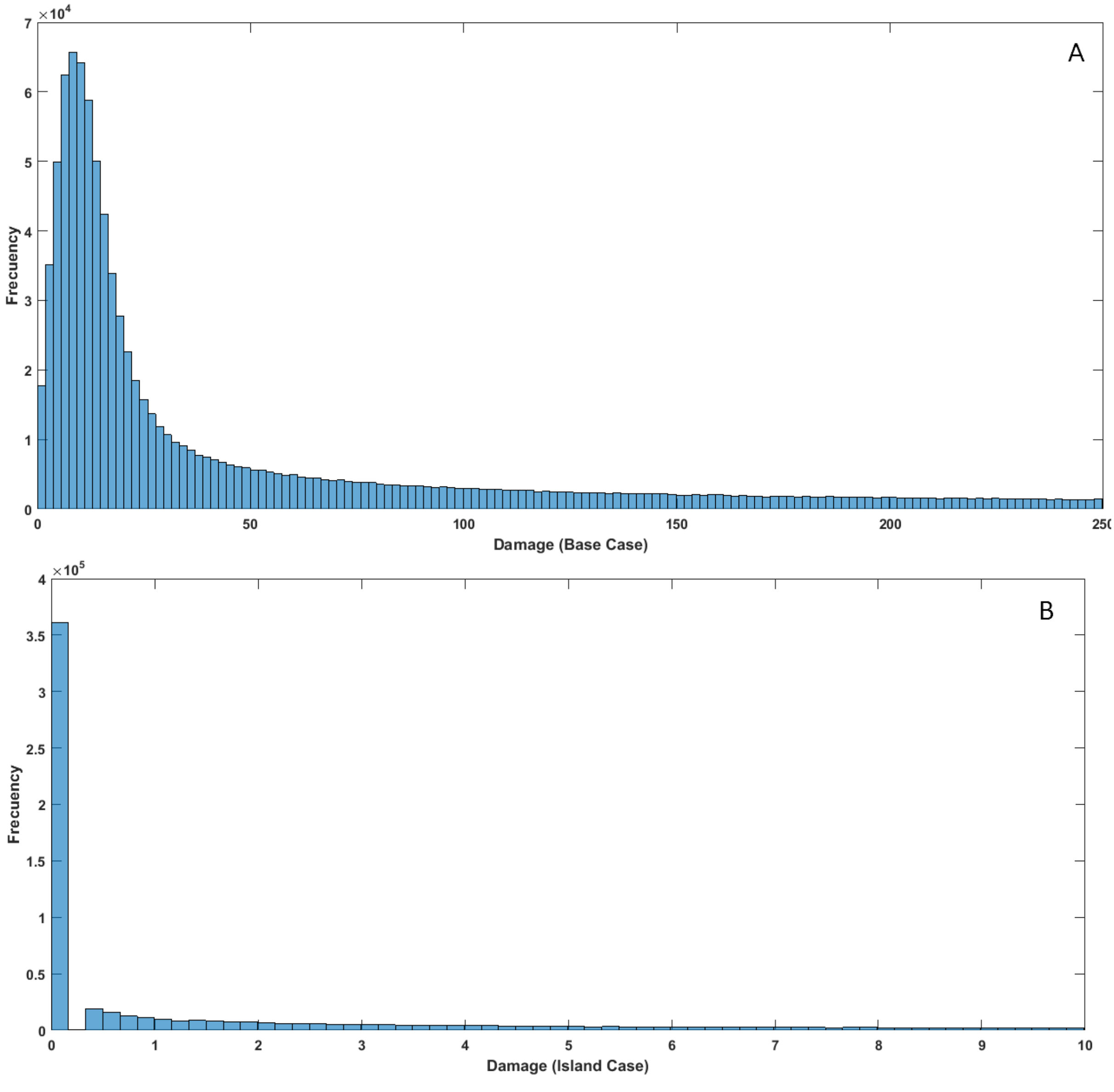


**Figure 3.** Probability distribution of damages as a function of the frequency of extreme events, for the case $\rho - \mu = 0.065$. Figure A represents the distribution in the baseline. Figure B represents the probability distribution once the opening of the canal is finished.

**Table 3.** Probabilities of 100 and 500-year flood events after the opening of the Deusto canal ( $T = 100$).

| No. 100-year floods (i=2) | No. 500-year floods (i=3) | Probability |
|---|---|---|
| 0 | 0 | 0.3606 |
| 1 | 0 | 0.3065 |
| 2 | 0 | 0.1303 |
| 3 | 0 | 0.0369 |
| 4 | 0 | 0.0078 |
| 5 | 0 | 0.0013 |
| 0 | 1 | 0.0613 |
| 1 | 1 | 0.0521 |
| 2 | 1 | 0.0221 |
| 3 | 1 | 0.0063 |
| 4 | 1 | 0.0013 |
| 0 | 2 | 0.0052 |
| 1 | 2 | 0.0044 |
| 2 | 2 | 0.0019 |
| 3 | 2 | 0.0005 |
| Other cases | | 0.0013 |
| **Total** | | **1.0000** |

**Table 4.** Measures of risk for the baseline and the adaptation scenario. Risk reduction is also shown, as the difference between the previous two situations. Damages are in millions of euros.

| | $\rho - \mu = 0.045$ | | | $\rho - \mu = 0.065$ | | |
|---|---|---|---|---|---|---|
| | mean | VaR(95%) | ES(95%) | mean | VaR(95%) | ES(95%) |
| A. Baseline (Zorrotzaurre is a peninsula) | 92.81 | 330.49 | 444.65 | 65.41 | 266.80 | 370.73 |
| B. Opening (Zorrotzaurre is an island) | 34.33 | 146.46 | 240.07 | 24.19 | 106.42 | 196.67 |
| Damage and risk reduction (A-B) | 58.47 | 184.02 | 204.57 | 41.21 | 160.39 | 174.06 |

If we compare these results with the values presented in Table 4 for avoided damages, we observe that the averages obtained via the Monte Carlo simulation differ slightly from the theoretical values. For example, for the case where $\rho - \mu = 0.045$, the mean avoided damage in Table 4 is 93.5 M EUR versus 92.81 M EUR presented above. Similar small differences are observed when $\rho - \mu = 0.065$ as well. This difference results from the use of two different methods, one numerical (MonteCarlo) and another one analytical (the Poisson distribution). Due to the number of simulations and the random numbers used with the MonteCarlo method, results slightly differ from those obtained through analytical methods. In turn, analytic methods such as the Poisson distribution are useful to estimate mean values, but not risk measures.

Figure 4 shows the reduction of risk, measured as ES(95%), for the baseline (peninsula) and canal opening (island) cases.

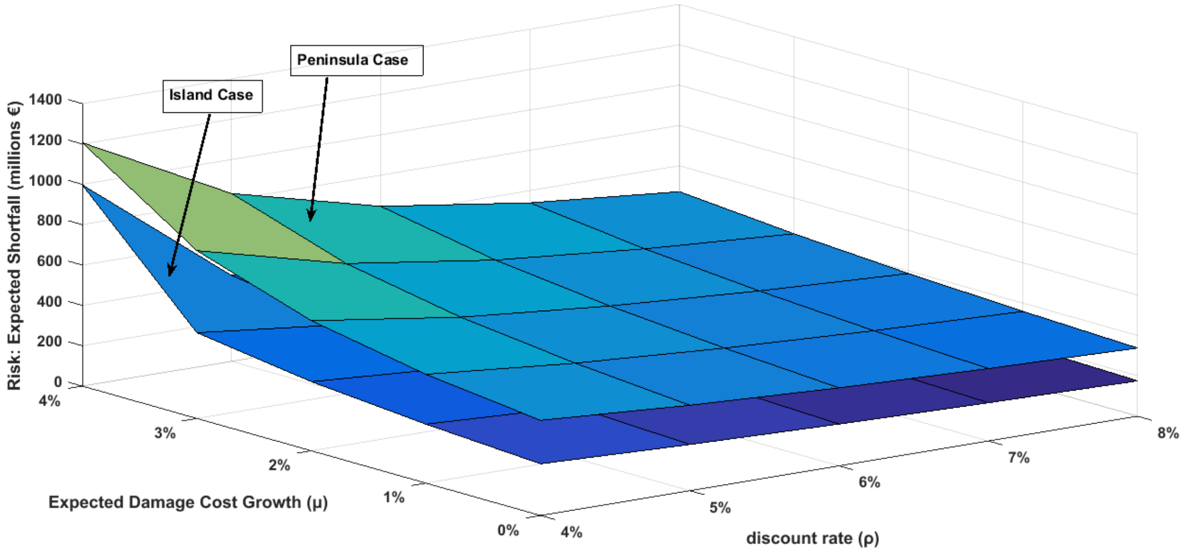



**Figure 4.** Representation of the Expected Shortfall (ES(95%) for the baseline and opening scenario as a function of $\rho-\mu$.

## 2.6 An assessment of risk with stochastic damage

The damage distributions generated so far are missing one important factor: damages do not increase deterministically over time but rather behave stochastically, though with an expected value identical to the case of deterministic growth. This means that damages resulting from a given flood are not fixed but can vary for a number of reasons. This being so, if flood events in each class *i* take place, the damage is given by $d_i(t)=d_i(0)S_t$, where $S_t$ is a variable that follows a stochastic process of the geometric Brownian motion (GBM) type, as given by equation (9):

$$dS_t = \mu S_t dt + \sigma\, S_t dW_t \ , \tag{9}$$

Where $S_i$ stands for the stochastic damage $S_0 = 1$. This present value increases at a rate $\mu$. Moreover, $\sigma$ is the instantaneous volatility, and $dW_t$ stands for the increment to a standard Wiener process.

The significant characteristics of this model include the fact that it does not generate negative values, so $S_t > 0$ at all times. At a time $t$ this distribution process generates a log-normal distribution (Hull, 2006; Abadie and Chamorro, 2013; Wilmott, 2014), where the first moment is:

$$E_0(d_i(t)) = E_0(d_i(0)S_t) = S_0 d_i(0)e^{\mu t} = d_i(0)e^{\mu t} \tag{10}$$

Table 5 shows damage estimates depending on volatility.

**Table 5.** Risk measures of damages taking account of volatility.

| Volatility | Case | $\rho-\mu=0.045$ | | | $\rho-\mu=0.065$ | | |
|---|---|---|---|---|---|---|---|
| | | Mean Montecarlo | Var(95%) | ES(95%) | Mean Montecarlo | Var(95%) | ES(95%) |
| σ=0 | A Base Line (Peninsula) | 92.81 | 330.49 | 444.65 | 65.41 | 266.80 | 370.73 |
| | B. Opening (Island) | 34.33 | 146.46 | 240.07 | 24.19 | 106.42 | 196.67 |
| | Damage risk reduction | 58.47 | 184.02 | 204.57 | 41.21 | 160.39 | 174.06 |
| σ=0.01 | A Base Line (Peninsula) | 92.80 | 330.66 | 445.14 | 65.40 | 266.93 | 370.98 |
| | B. Opening (Island) | 34.33 | 146.55 | 240.22 | 24.19 | 106.36 | 196.74 |
| | Damage risk reduction | 58.47 | 184.10 | 204.91 | 41.21 | 160.57 | 174.25 |
| σ=0.02 | A Base Line (Peninsula) | 92.80 | 331.24 | 446.45 | 65.40 | 267.38 | 371.69 |
| | B. Opening (Island) | 34.33 | 146.66 | 240.72 | 24.19 | 106.33 | 196.97 |
| | Damage risk reduction | 58.47 | 184.58 | 205.74 | 41.21 | 161.04 | 174.72 |
| σ=0.03 | A Base Line (Peninsula) | 92.79 | 332.41 | 448.61 | 65.39 | 268.03 | 372.84 |
| | B. Opening (Island) | 34.32 | 147.04 | 241.54 | 24.19 | 106.36 | 197.37 |
| | Damage risk reduction | 58.47 | 185.37 | 207.07 | 41.21 | 161.67 | 175.46 |
| σ=0.04 | A Base Line (Peninsula) | 92.79 | 333.83 | 451.64 | 65.39 | 268.93 | 374.43 |
| | B. Opening (Island) | 34.32 | 147.39 | 242.72 | 24.18 | 106.56 | 197.95 |
| | Damage risk reduction | 58.47 | 186.44 | 208.92 | 41.20 | 162.37 | 176.48 |
| σ=0.05 | A Base Line (Peninsula) | 92.79 | 335.63 | 455.58 | 65.38 | 270.03 | 376.51 |
| | B. Opening (Island) | 34.32 | 147.74 | 244.25 | 24.18 | 106.68 | 198.70 |
| | Damage risk reduction | 58.46 | 187.90 | 211.32 | 41.20 | 163.34 | 177.81 |

The results presented in Table 5 show that the expected values are the same as in the case of deterministic growth in damage. However, the risks now grow as volatility increases. In other words, risk *R* is a function of $\rho-\mu$ and $\sigma$, i.e. $R(\rho-\mu,\sigma)$.

## 2.7 Options to Invest and Applying ROA to the Data

In this section we develop an application of ROA to this project, assuming that there is an option to invest in adaptation by opening up the Deusto canal and turning the Zorrozaure peninsula into an island in period $T$, for which investment costs of $I$ must be paid. If investment is made at time $t$, there is an immediate present value of damage avoided for a period of 85 years $[t,85]$ given equation (11).

$$E_t(D_P^{t,85})-E_t(D_I^{t,85})=S_T\,\frac{(d_1^P-d_1^I)\lambda_1+(d_2^P-d_2^I)\lambda_2+(d_3^P-d_3^I)\lambda_3}{\rho-\mu}[1-e^{-(85-t)(\rho-\mu)}] \tag{11}$$

where the scripts $P$ and $I$ refer to the cases of the baseline (*peninsula, P*) and the opening of the canal (*island, I*), respectively, and where the remaining useful lifetime is $85 - t$ if investment is made at time $t$.

At time $T$ the decision made will be to invest if the expected present value of the avoided damage over the course of the 85 years is greater than the cost of investment, i.e. if equation (12) is met:

$$W_T = (S_T \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3)}{\rho - \mu}[1 - e^{-(85-T)(\rho-\mu)}] - I) > 0$$

(12)

If investment is moved to an earlier time $t$ for an interval $\Delta t$ several things may happen:

a. During that period there may be damage with an expected present value as follows:

$$E_t(D^{t,t+\Delta t}) = S_t \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3}{\rho - \mu}[1 - e^{-(\rho-\mu)\Delta t}]$$
$$\approx S_t((d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3)\Delta t$$

(13)

b. But the continuation value obtained will be the following:

$$W_t = (pW^+ + (1-p)W^-)e^{-\rho\Delta t}) \qquad (14)$$

Where the values $W^+$ and $W^-$ are the valuations of the nodes where $S$ increases and decreases respectively. The valuation at an intermediate node of a binomial tree (see Figure 5) is therefore:

$$W_t = \max(S_t \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3}{\rho - \mu}[1 - e^{-(85-t)(\rho-\mu)}] - I);$$
$$(pW^+ + (1-p)W^-)e^{-\rho\Delta t})) - S_t \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3}{\rho - \mu}[1 - e^{-(\rho-\mu)\Delta t}])$$

(15)

With this it is possible to build a binomial tree, given the values of σ, μ and ρ. A ”binomial model assumes than the underlying damage growt ($S_t$) follows a binomial process, tha is at any time the dramage growth can only change to one of two possible values. Under this assumption the asset price has a binomial distribution” (Clewlow and Strickland, 1998: 10).

At the outset the value of immediate investment is given by (16):

$$NPV_0 = \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3}{\rho - \mu}[1 - e^{-85(\rho-\mu)}] - I \qquad (16)$$

And the continuation value is given by equation (17):

$$C_0 = (pW^+ + (1-p)W^-)e^{-\rho\Delta t})) - S_t \frac{(d_1^P - d_1^I)\lambda_1 + (d_2^P - d_2^I)\lambda_2 + (d_3^P - d_3^I)\lambda_3}{\rho - \mu}[1 - e^{-(\rho-\mu)\Delta t}] \tag{17}$$

The values of $I$ that give $NPV_0 = C_0$ define the optimal exercise boundary -i.e. the cost of investment at which the investor is indifferent between investing now and waiting to invest in a future period.

Initially we develop the calculations for a case in which $\sigma = 0.05;\ \rho = 0.08;\ \mu = 0.015;\ T = 1;\ \Delta t = 0.5,;\ I = 12.1$. This means building a tree with only two steps in which the investment option is only available for one year. Initially $S_0 = 1$. After one step $\Delta t = 0.5$ the value of $S$ becomes $uS$ if it increases with probability $p_u$ or $dS$ if it decreases with probability $p_d$, where

$$u = e^{\sigma\sqrt{\Delta t}} = 1.0360\ ;\ d = e^{-\sigma\sqrt{\Delta t}} = 0.9653 \tag{18}$$

$$p_u = \frac{e^{\mu\Delta t} - d}{u - d} = 0.5976\ ;\ p_d = (1 - p_u) = 0.4024 \tag{19}$$

Figure 5 presents the results for two investment options. The left chart has been estimated for an initial investment of 12.1 M EUR, which is the actual cost of opening the canal. The chart on the right shows the results that determine the boundary value of investment cost $I$ between the “investment region” and the “wait region”. This occurs for $I^* = 43.60$, which represents the investment values for which the net present value of investment equals the wait value.

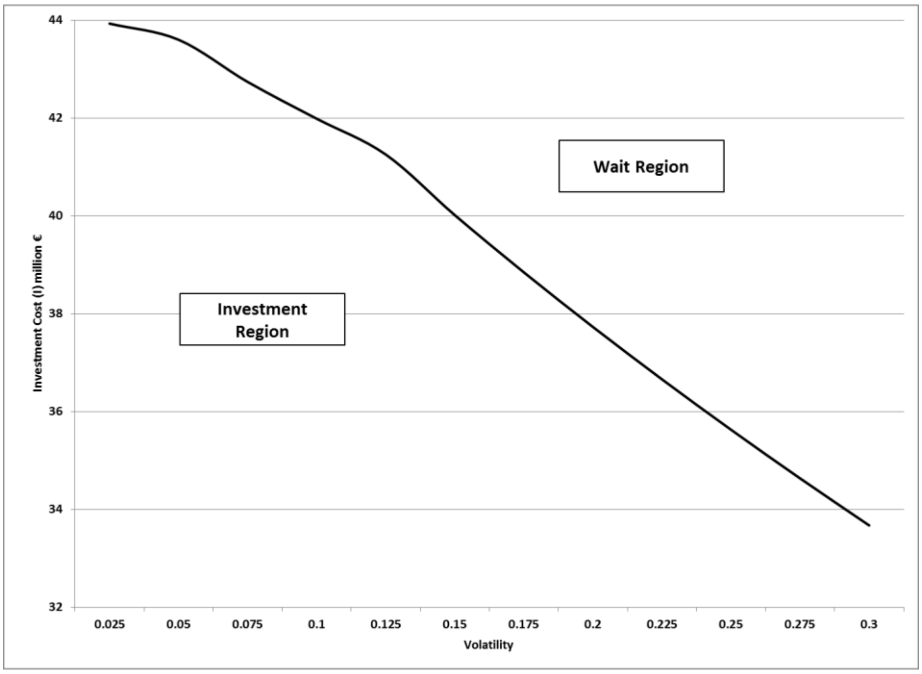
There are various parameters which influence the maximum cost that can be accepted for making investment immediately. For example, volatility can change the boundary of the wait-investment regions. As shown in Table 6, the greater the volatility, the lower the investment boundary. In other words, greater volatility makes potential investors more demanding and they invest only when the cost is lower.

**Table 6.** Changes in investment boundaries depending on volatility.

| **Volatility $\sigma$** | **Investment $I^*$** |
|---|---|
| 0.05 | 49.60 |
| 0.10 | 41.98 |
| 0.20 | 37.73 |
| 0.30 | 33.67 |

If the discount rate $\rho$ increases, both the net present value and the continuation value decrease, but the first more than the second, therefore $I^*$ becomes smaller. For the baseline scenario, with $\rho = 0.10$ the figure that results is $I^* = 34.28$. If damage grows as a result of climate change and/or socio-economic development, $\mu$ increases. For the baseline scenario and for $\mu = 0.035$ then the resulting figure is $I^* = 59.22$. The increase in $\mu$ results in higher investment costs being accepted.

Now consider a more elaborated case in which $T = 10$ years and $\Delta t = 1/50$. A binomial tree with 500 steps then needs to be built (Figure 5). In this case $I^* = 48.37$. The value is lower here because the period in which the option can be exercised is longer.




An optimal exercise boundary graph can also be drawn up depending on volatility (Figure 6). An increase in volatility, and therefore in uncertainty, results in investment being made immediately only if the cost is substantially lower. This example has been defined for a twenty-year period in which the option can be exercised.

**Figure 5.** Investment option tree (data are presented in millions of euros).

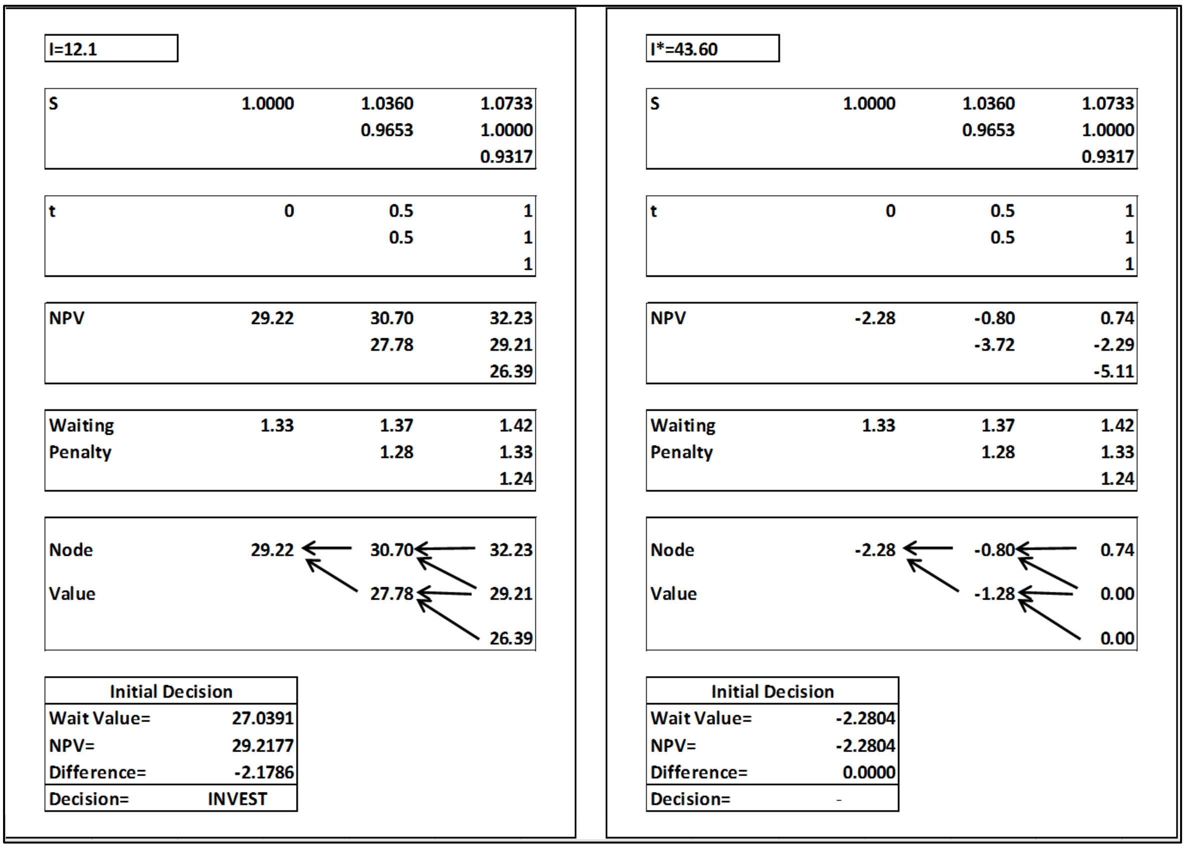



**Figure 6.** Investment and Wait Regions depending on volatility

## 2.8 Conclusions on the Use of ROA

The example provided here shows the information needed to undertake a proper ROA. In a number of examples in the climate literature the authors use the language of ROA but they do not really apply the method fully.

In making an application we have also shown that it is relatively complex to use the tool. The risk estimation can be carried out using tools that are now available but in the climate adaptation context ROA cannot generally be applied through financial instruments, as it can in the financial context. The example given here of flood prevention is a public good, which holds for most public adaptation projects. Decisions of timing can be informed by using decision trees, in which the concepts of value at risk and expected loss play a central role. But the decision points remain arbitrary and including many of them generates very complex decision trees, which are difficult to communicate to policy-makers.

In our opinion ROA is useful as a guide to some of the key parameters in public-sector decision-making for adaptation but it is not really suitable for use as a standard tool that can be applied in a straightforward manner.

# 3 Application of Acceptable Risk

As noted the method outlined above is complicated and not straightforward to apply. Here we consider another way in which the timing of investments can be determined using similar information on value at risk (VAR) and expected shortfall. The method is applied to sea level rise (SLR) resulting from climate change and consists of estimating the expected damage distribution from SLR as described in the previous section. This is applied to a city at different points in time. The government then decides on an acceptable level of risk and the year for adaptation action is obtained by looking for the moment in time when the expected shortfall (i.e. expected damage from events where the probability of their occurrence (1-α) constitutes the cases where damages exceeds VAR(α)) is greater the acceptable damage. The latter may be set at (0.1%, 0.5% and 1% of a city's GDP). This way we obtain the exact year when each city needs to start adapting to climate change.

## 3.1 Outline of the Statistical Framework and Data Analysed

The statistical framework is similar to that used for ROA analysis. First we estimate the total damage costs for the major coastal cities using a continuous stochastic Geometric Brownian Motion (GBM) model to understand the probability distribution of SLR in each moment in time.

The GBM model was described in Section 2.1.5 above. For SLR it was calibrated using the data from the latest IPCC scenarios, RCP 2.6, 4.5, 6.0 and 8.5 (Church et al. 2013). This allows a SLR distribution function that is log-normal at all times, so there were no negative values. The expected SLR drift is obtained by minimising the sum of the square of the differences with the theoretical values. The volatility is calculated using 1,000,000 Monte Carlo simulations to approximate the theoretical percentile 95 % distribution of SLR at 2100 for each IPCC scenario in the calibration process (see annex for further details).

## 3.2 Results Obtained

The mean losses by 2050 taking uncertainty into consideration ranged from US$1,108 billion to US$1,704 billion, depending on the reference used, in line with previous results from Hallegatte et al. (2013). Table 7 shows the mean annual economic loss estimates for the top 20 cities (left column) and a comparison with earlier estimates (right column). Eighteen out of the top 20 cities in the ranking are the same, although the damage costs vary in our model depending on the IPCC scenario. The losses per scenario average 20% but in some cases may be as high as 50% or even 68%. Two of the most affected-for-extreme-events US cities are in the top 20: New Orleans and New York.

**Table 7.** Cities ranked by annual average losses (AAL) in 2050.

| AAL damage costs (US$ million) by IPCC RCPs | | | | | | AAL damage costs (US$ million) for an optimistic sea-level rise scenario from Hallegatte et al. (2013) | | |
|---|---|---|---|---|---|---|---|---|
| | Urban Agglomeration | RCP2.6 | RCP4.5 | RCP6.0 | RCP8.5 | | Urban Agglomeration | SLR 20 cm |
| 1 | Guangzhou (S) | 239,465 | 255,544 | 259,273 | 302,111 | 1 | Guangzhou (S) | 254.721 |
| 2 | New Orleans (S) | 141,464 | 154,962 | 157,833 | 183,743 | 2 | New Orleans (S) | 161.141 |
| 3 | Mumbai | 97,429 | 126,163 | 132,511 | 194,011 | 3 | Mumbai | 107.285 |
| 4 | Osaka-Kobe (S) | 75,456 | 81,781 | 83,116 | 94,831 | 4 | Osaka-Kobe (S) | 84.968 |
| 5 | Tokyo (S) | 53,918 | 59,012 | 60,080 | 69,281 | 5 | Tokyo (S) | 61.737 |
| 6 | Nagoya (S) | 51,056 | 55,389 | 56,288 | 63,754 | 6 | Nagoya (S) | 57.954 |
| 7 | Kolkata (S) | 50,386 | 55,658 | 56,845 | 69,550 | 7 | Kolkata (S) | 56.303 |
| 8 | Tianjin (S) | 36,968 | 40,174 | 40,899 | 48,734 | 8 | Tianjin (S) | 40.492 |
| 9 | Al-Iskandariyah(S) | 31,791 | 36,174 | 37,240 | 50,861 | 9 | Al-Iskandariyah (S) | 34.621 |
| 10 | Guayaquil (S) | 29,604 | 33,510 | 34,493 | 47,859 | 10 | Guayaquil (S) | 31.288 |
| 11 | Shenzen | 20,623 | 29,689 | 31,930 | 58,039 | 11 | Krung_Thep (Bangkok) (S) | 20.778 |
| 12 | Fukuoka-Kitakyushu (S) | 19,199 | 19,975 | 20,157 | 22,272 | 12 | Fukuoka-Kitakyushu (S) | 19.904 |
| 13 | Krung_Thep (Bangkok) (S) | 19,129 | 22,277 | 23,058 | 33,403 | 13 | Vancouver (S) | 18.912 |
| 14 | Vancouver (S) | 17,988 | 18,900 | 19,109 | 21,454 | 14 | Shenzen | 17.553 |
| 15 | Zhanjiang (S) | 15,266 | 16,527 | 16,810 | 19,807 | 15 | Zhanjiang (S) | 16.709 |
| 16 | Jakarta (S) | 15,179 | 16,543 | 16,864 | 20,674 | 16 | Jakarta (S) | 16.354 |
| 17 | Xiamen (S) | 11,001 | 12,030 | 12,261 | 14,703 | 17 | Xiamen (S) | 12.182 |
| 18 | Abidjan | 9,924 | 14,162 | 15,191 | 26,877 | 18 | Hiroshima (S) | 9.456 |
| 19 | Hiroshima (S) | 9,003 | 9,441 | 9,541 | 10,654 | 19 | Los Angeles-Long Beach Santa Ana | 9.427 |
| 20 | New York-Newark | 8,440 | 11,109 | 11,753 | 19,027 | 20 | Surat | 9.070 |

* (S) indicates that the city is subject to significant subsidence. The reference data are taken from Hallegatte et al. (2013).

To move on to incorporating risk in the calculation, we used 1,000,000 simulated SLR values for each scenario and time *t*. Then we add to the SLR function the expected subsidence level, which is specific for each city (Hallegatte et al. 2013). Each SLR for each city and time *t* causes a specific damage cost. As we have developed a damage distribution, we can also calculate the 95% percentile VaR (95%), we have 50,000 values for the damages of the most unfavourable situations, which enables us to obtain highly accurate values of ES (95%).  VaR (95%) is the value of the loss corresponding to the damage function in the 95% percentile. ES (95%) refers to the mean expected loss when the value VaR is exceeded. This model considers α=95%, but another values of α could be used.

We then define a level of risk at 0.5% of each city's GDP in 2020. The same level of risk is used in 2050. This enables us to estimate, for a desired level of risk, the optimal size of the defences in cm needed in 2020 and 2050 to avoid water overflowing. This can be calculated for any given year as we have estimated a continuous damage function. The years for adaptation action are obtained by looking for the moment in time when ES (95%) exceeds the maximum acceptable damage (0.1%, 0.5% and 1% of each city's GDP in this example). This way we obtained the exact year when each city needs to start adapting to climate change.

The values for the top 20 cities are shown in Table 8 for the four IPCC scenarios. The 95% percentile represents the low-probability but high-damage impacts repeatedly discussed earlier in climate change economics literature (Weitzman 2007; Weitzman 2009; Nordhaus 2011; Weitzman 2013). These low-probability events are very important due to the huge magnitude of the potential damage (Pindyck 2011). Our calculations show that the mean values for the worst 95% cases in 2050 can be up to 300% higher and at least 118% higher than average damage.

**Table 8.** Cities ranked by annual average losses (AAL) in 2050, considering the 95 percentiles of Expected Shortfall (ES) and Value at Risk (VaR).

| Urban Agglomeration | RCP2.6 | | RCP4.5 | | RCP6.0 | | RCP8.5 | |
|---|---|---|---|---|---|---|---|---|
| | VaR(95%) | ES(95%) | VaR(95%) | ES(95%) | VaR(95%) | ES(95%) | VaR(95%) | ES(95%) |
| 1 Guangzhou_Guangdong (S) | 288,144 | 308,315 | 314,486 | 334,608 | 319,314 | 339,178 | 384,333 | 408,405 |
| 2 Mumbai (Bombay) | 179,105 | 204,234 | 211,921 | 236,987 | 217,935 | 242,680 | 298,931 | 328,918 |
| 3 New Orleans (S) | 177,954 | 188,101 | 191,205 | 201,327 | 193,633 | 203,626 | 226,340 | 238,449 |
| 4 Osaka-Kobe (S) | 92,370 | 96,837 | 98,204 | 102,660 | 99,273 | 103,672 | 113,672 | 119,003 |
| 5 Kolkata (Calcutta) (S) | 65,762 | 71,471 | 73,218 | 78,912 | 74,584 | 80,206 | 92,985 | 99,798 |
| 6 Tokyo (S) | 67,437 | 70,876 | 71,929 | 75,360 | 72,752 | 76,139 | 83,839 | 87,944 |
| 7 Nagoya (S) | 62,398 | 65,080 | 65,901 | 68,577 | 66,543 | 69,185 | 75,189 | 78,390 |
| 8 Shenzen | 50,717 | 62,321 | 65,871 | 77,445 | 68,648 | 80,074 | 106,049 | 119,896 |
| 9 Al-Iskandariyah (Alexandria) (S) | 45,906 | 52,717 | 54,800 | 61,594 | 56,430 | 63,137 | 78,383 | 86,511 |
| 10 Tianjin (S) | 46,366 | 49,911 | 50,995 | 54,532 | 51,844 | 55,335 | 63,271 | 67,501 |
| 11 Guayaquil (S) | 42,719 | 49,617 | 51,728 | 58,609 | 53,379 | 60,173 | 75,615 | 83,848 |
| 12 Krung_Thep (Bangkok) (S) | 29,512 | 34,783 | 36,396 | 41,654 | 37,657 | 42,848 | 54,648 | 60,938 |
| 13 Abidjan | 23,657 | 28,797 | 30,369 | 35,496 | 31,599 | 36,660 | 48,165 | 54,298 |
| 14 Dubayy (Dubai) | 18,220 | 23,133 | 24,635 | 29,535 | 25,811 | 30,648 | 41,645 | 47,507 |
| 15 Fukuoka-Kitakyushu (S) | 21,570 | 22,576 | 22,883 | 23,887 | 23,124 | 24,114 | 26,366 | 27,566 |
| 16 Vancouver (S) | 20,712 | 21,799 | 22,132 | 23,216 | 22,392 | 23,462 | 25,895 | 27,192 |
| 17 Jakarta (S) | 19,386 | 21,216 | 21,775 | 23,601 | 22,213 | 24,015 | 28,111 | 30,295 |
| 18 Zhanjiang (S) | 18,925 | 20,263 | 20,672 | 22,006 | 20,992 | 22,309 | 25,303 | 26,899 |
| 19 New York-Newark | 17,032 | 20,222 | 21,198 | 24,380 | 21,962 | 25,103 | 32,244 | 36,051 |
| 20 Boston | 15,859 | 19,464 | 20,567 | 24,163 | 21,430 | 24,979 | 33,049 | 37,350 |

* S indicates that the city is subject to significant subsidence. The reference data are taken from Hallegatte et al. (2013).
** The values for the remaining cities are presented in Supplementary Table 1. Together with VaR and ES(95%), the supp. table includes mean values, such as those included in Table 1.

Finally we propose a methodology for controlling the risk of SLR. We do this by limiting the mean annual loss in 2020 to 0.5% of the GDP of the cities analysed. This is proposed as a threshold of acceptable risk. The values fall within the boundaries of the estimated investment needs for adaptation (UNFCCC 2007) and annual adaptation costs of SLR (Agrawala et al. 2011). Any other limit could be used, including city-specific values.

Although some cities such as Tokyo, Krung Thep (Bangkok), Dubayy (Dubai), New York and Boston have very limited adaptation needs (less than 5 cm) in 2020, these demands increase exponentially by 2050. Thirteen of the top 20 cities need more than half a meter of protection by 2050 (Table 9). Guangzhou, New Orleans, Kolkata, Guayaquil and Zhanjiang need more than 80cm in RCP 8.5.

**Table 9.** Optimal size of flood defences (cm) in 2020 and 2050 in order for ES (95%) values to be 0.5% of the city GDP or less.

| Urban Agglomeration | 2020 | | | | 2050 | | | |
|---|---|---|---|---|---|---|---|---|
| | **RCP2.6** | **RCP4.5** | **RCP6.0** | **RCP8.5** | **RCP2.6** | **RCP4.5** | **RCP6.0** | **RCP8.5** |
| 1 Guangzhou_Guangdong (S) | 25.8 | 26.3 | 26.3 | 28.1 | 70.2 | 72.92 | 73.4 | 81.3 |
| 2 Mumbai (Bombay) | 10.7 | 11.2 | 11.7 | 12.9 | 32.9 | 35.7 | 36.2 | 43.5 |
| 3 New Orleans (S) | 26.2 | 26.7 | 26.8 | 28.5 | 69.5 | 72.3 | 72.7 | 80.4 |
| 4 Osaka-Kobe (S) | 19.9 | 20.4 | 20.5 | 22.2 | 60.8 | 63.6 | 64.1 | 71.4 |
| 5 Kolkata (Calcutta) (S) | 19.6 | 20.1 | 20.1 | 21.8 | 72.6 | 75.4 | 75.8 | 83.1 |
| 6 Tokyo (S) | 0.0 | 0.0 | 0.0 | 0.0 | 33.7 | 36.5 | 37.0 | 44.3 |
| 7 Nagoya (S) | 23.5 | 24.0 | 24.1 | 25.8 | 64.4 | 67.2 | 68.3 | 75.6 |
| 8 Shenzen | 9.8 | 10.3 | 10.4 | 12.1 | 36.7 | 39.5 | 40.0 | 47.3 |
| 9 Al-Iskandariyah (Alexandria) (S) | 23.5 | 24.0 | 24.1 | 25.8 | 65.1 | 67.9 | 68.4 | 75.8 |
| 10 Tianjin (S) | 23.0 | 23.5 | 23.6 | 25.3 | 67.7 | 70.4 | 70.9 | 78.2 |
| 11 Guayaquil (S) | 25.9 | 26.5 | 26.5 | 28.2 | 77.5 | 80.3 | 80.7 | 88.3 |
| 12 Krung_Thep (Bangkok) (S) | 1.1 | 1.7 | 1.7 | 3.4 | 54.1 | 56.9 | 57.4 | 64.7 |
| 13 Abidjan | 11.7 | 12.2 | 12.3 | 14.0 | 36.1 | 38.9 | 39.3 | 46.9 |
| 14 Dubayy (Dubai) | 2.8 | 3.3 | 3.4 | 5.1 | 24.7 | 27.4 | 27.9 | 35.4 |
| 15 Fukuoka-Kitakyushu (S) | 23.8 | 24.4 | 24.4 | 26.1 | 64.8 | 67.6 | 68.1 | 75.6 |
| 16 Vancouver (S) | 23.7 | 24.3 | 24.3 | 26.0 | 64.8 | 67.7 | 68.2 | 75.6 |
| 17 Jakarta (S) | 20.1 | 20.7 | 20.7 | 22.4 | 66.5 | 69.3 | 69.8 | 77.1 |
| 18 Zhanjiang (S) | 24.4 | 25.0 | 25.5 | 26.7 | 69.7 | 72.5 | 72.9 | 80.3 |
| 19 New York-Newark | 0.0 | 0.0 | 0.0 | 0.0 | 12.1 | 14.9 | 15.4 | 22.8 |
| 20 Boston | 0.7 | 1.2 | 1.3 | 3.0 | 20.5 | 23.3 | 23.7 | 31.3 |

* (S) indicates that the city is subject to significant subsidence. The reference data are taken from Hallegatte et al. (2013).
** The values for the rest of the cities are presented in Supplementary Table 2 (a, b).

Note that the top 20 ranking cities with the highest ES values differs from the top 20 of cities with the greatest need for adaptation (Table 10 and 11). This is due to important differences in the expected economic growth rates of each city and the fact that acceptable risk level is set in terms of GDP. We consider 2020 and 2050 but corresponding figures can be calculated for any given year (Supplementary material). In this ranking three cities need more than 90 cm of adaptation: Khulna, Hai Phòng and Thành-Pho-Ho-Chí-Minh respectively. This last city in fact needs more than 100cm. Overall adaptation needs increase dramatically after 2050 as SLR is much more acute.

**Table 10.** Level of protection (in cm) needed by 2020 and 2050. Risk measured as 0,5% of cities' GDP in 2020. Ranking based on ES(95%) values.

| City ID | Urban Agglomeration | 2020 | | | | 2050 | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | RCP2.6 | RCP4.5 | RCP6.0 | RCP8.5 | RCP2.6 | RCP4.5 | RCP6.0 | RCP8.5 |
| 40 | Guangzhou_Guangdong | 25,8 | 26,3 | 26,3 | 28,1 | 70,2 | 76,6 | 73,4 | 81,3 |
| 78 | Mumbai (Bombay) | 10,7 | 11,2 | 11,7 | 12,9 | 32,9 | 35,7 | 36,2 | 43,5 |
| 84 | New Orleans | 26,2 | 26,7 | 26,8 | 28,5 | 69,5 | 72,3 | 72,7 | 80,4 |
| 88 | Osaka-Kobe | 19,9 | 20,4 | 20,5 | 22,2 | 60,8 | 63,6 | 64,1 | 71,4 |
| 58 | Kolkata (Calcutta) | 19,6 | 20,1 | 20,1 | 21,8 | 72,6 | 75,4 | 75,8 | 83,1 |
| 126 | Tokyo | 0,0 | 0,0 | 0,0 | 0,0 | 33,7 | 36,5 | 37,0 | 44,3 |
| 80 | Nagoya | 23,5 | 24,0 | 24,1 | 25,8 | 64,4 | 67,2 | 68,3 | 75,6 |
| 114 | Shenzen | 9,8 | 10,3 | 10,4 | 12,1 | 36,7 | 39,5 | 40,0 | 47,3 |
| 5 | Al-Iskandariyah (Alexandria) | 23,5 | 24,0 | 24,1 | 25,8 | 65,1 | 67,9 | 68,4 | 75,8 |
| 125 | Tianjin | 23,0 | 23,5 | 23,6 | 25,3 | 67,7 | 70,4 | 70,9 | 78,2 |
| 41 | Guayaquil | 25,9 | 26,5 | 26,5 | 28,2 | 77,5 | 80,3 | 80,7 | 88,3 |
| 59 | Krung_Thep (Bangkok) | 1,1 | 1,7 | 1,7 | 3,4 | 54,1 | 56,9 | 57,4 | 64,7 |
| 1 | Abidjan | 11,7 | 12,2 | 12,3 | 14,0 | 36,1 | 38,9 | 39,3 | 46,9 |
| 31 | Dubayy (Dubai) | 2,8 | 3,3 | 3,4 | 5,1 | 24,7 | 27,4 | 27,9 | 35,4 |
| 36 | Fukuoka-Kitakyushu | 23,8 | 24,4 | 24,4 | 26,1 | 64,8 | 67,6 | 68,1 | 75,6 |
| 129 | Vancouver | 23,7 | 24,3 | 24,3 | 26,0 | 64,8 | 67,7 | 68,2 | 75,6 |
| 52 | Jakarta | 20,1 | 20,7 | 20,7 | 22,4 | 66,5 | 69,3 | 69,8 | 77,1 |
| 136 | Zhanjiang | 24,4 | 25,0 | 25,5 | 26,7 | 69,7 | 72,5 | 72,9 | 80,3 |
| 85 | New York-Newark | 0,0 | 0,0 | 0,0 | 0,0 | 12,1 | 14,9 | 15,4 | 22,8 |
| 16 | Boston | 0,7 | 1,2 | 1,3 | 3,0 | 20,5 | 23,3 | 23,7 | 31,3 |

**Table 11.** Level of protection (in cm) needed by 2020 and 2050. Risk measured as 0,5% of cities' GDP in 2020. Ranking based on ES(95%) values.

| City ID | Urban Agglomeration | 2020 | | | | 2050 | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | RCP 2.6 | RCP 4.5 | RCP 6.0 | RCP 8.5 | RCP 2.6 | RCP 4.5 | RCP 6.0 | RCP 8.5 |
| 124 | Thành-Pho-Ho-Chí-Minh (Ho Chi Minh City) | 31,1 | 31,6 | 31,7 | 33,4 | 133,0 | 135,8 | 136,3 | 143,6 |
| 84 | New Orleans | 26,2 | 26,7 | 26,8 | 28,5 | 69,5 | 72,3 | 72,7 | 80,4 |
| 41 | Guayaquil | 25,9 | 26,5 | 26,5 | 28,2 | 77,5 | 80,3 | 80,7 | 88,3 |
| 40 | Guangzhou_Guangdong | 25,8 | 26,3 | 26,3 | 28,1 | 70,2 | 76,6 | 73,4 | 81,3 |
| 136 | Zhanjiang | 24,4 | 25,0 | 25,5 | 26,7 | 69,7 | 72,5 | 72,9 | 80,3 |
| 89 | Palembang | 24,2 | 24,7 | 24,9 | 26,4 | 71,6 | 74,4 | 74,9 | 82,2 |
| 55 | Khulna | 24,1 | 24,7 | 24,9 | 26,4 | 80,0 | 82,7 | 83,2 | 90,5 |
| 36 | Fukuoka-Kitakyushu | 23,8 | 24,4 | 24,4 | 26,1 | 64,8 | 67,6 | 68,1 | 75,6 |
| 129 | Vancouver | 23,7 | 24,3 | 24,3 | 26,0 | 64,8 | 67,7 | 68,2 | 75,6 |
| 5 | Al-Iskandariyah (Alexandria) | 23,5 | 24,0 | 24,1 | 25,8 | 65,1 | 67,9 | 68,4 | 75,8 |
| 80 | Nagoya | 23,5 | 24,0 | 24,1 | 25,8 | 64,4 | 67,2 | 68,3 | 75,6 |
| 125 | Tianjin | 23,0 | 23,5 | 23,6 | 25,3 | 67,7 | 70,4 | 70,9 | 78,2 |
| 46 | Hiroshima | 22,8 | 23,3 | 23,4 | 25,1 | 63,6 | 66,4 | 66,9 | 74,2 |
| 42 | Hai Phòng | 22,8 | 23,3 | 23,4 | 25,1 | 82,4 | 85,2 | 85,6 | 92,9 |
| 134 | Xiamen | 21,7 | 22,2 | 22,3 | 24,0 | 68,3 | 71,1 | 71,6 | 78,9 |
| 120 | Taipei | 20,9 | 21,5 | 21,5 | 23,2 | 63,7 | 66,6 | 67,1 | 74,4 |
| 86 | Ningbo | 20,6 | 21,2 | 21,2 | 22,9 | 63,5 | 66,2 | 66,7 | 74,0 |
| 52 | Jakarta | 20,1 | 20,7 | 20,7 | 22,4 | 66,5 | 69,3 | 69,8 | 77,1 |
| 88 | Osaka-Kobe | 19,9 | 20,4 | 20,5 | 22,2 | 60,8 | 63,6 | 64,1 | 71,4 |
| 58 | Kolkata (Calcutta) | 19,6 | 20,1 | 20,1 | 21,8 | 72,6 | 75,4 | 75,8 | 83,1 |

In planning adaptation, it is highly important to know the right time to start to build defences. Previous works have suggested the use of adaptive policies that can be changed or adjusted in time (see, for example, Kwakkel et al. 2015). Precisely, the method presented here can be used to design progressive adaptation strategies, which is achieved by analysing how ES changes every 5 years. Building up flexibility in decision-making for planning adaptation infrastructures is a great challenge but it can be achieved through this method as it will allow updating information as soon as assessments to show which IPCC scenario are closer to reality become available (IPCC 2014). All the cities in the top 20 ranking should undertake adaptation before 2020 (many of them should have started already) in all four IPCC scenarios and for three levels of risk (Methods), except for New York, Boston and Tokyo, which could wait a bit longer (Table 12) under some scenarios.

**Table 12.** Year in which cities should start adaptation based on RCP scenarios and different threshold of acceptable risk.

| **Urban agglomeration** | **0.1% damage** | | | | **0.5% damage** | | | | **1% damage** | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 2.6 | 4.5 | 6.0 | 8.5 | 2.6 | 4.5 | 6.0 | 8.5 | 2.6 | 4.5 | 6.0 | 8.5 |
| | Year to start adaptation | | | | Year to start adaptation | | | | Year to start adaptation | | | |
| Abidjan | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Al-Iskandariyah (Alexandria) | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Boston | 2010 | 2010 | 2010 | 2010 | 2015 | 2015 | 2015 | 2015 | 2025 | 2020 | 2020 | 2020 |
| Dubayy (Dubai) | 2010 | 2010 | 2010 | 2010 | 2015 | 2015 | 2015 | 2010 | 2015 | 2015 | 2015 | 2015 |
| Fukuoka-Kitakyushu | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Guangzhou_Guangdong | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Guayaquil | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Hai Phòng | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Hiroshima | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Jakarta | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Khulna | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Kolkata (Calcutta) | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Krung_Thep (Bangkok) | 2010 | 2010 | 2010 | 2010 | 2015 | 2015 | 2015 | 2015 | 2020 | 2020 | 2020 | 2020 |
| Mumbai (Bombay) | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Nagoya | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| New Orleans | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| New York-Newark | 2010 | 2010 | 2010 | 2010 | 2030 | 2025 | 2025 | 2025 | 2040 | 2035 | 2035 | 2030 |
| Ningbo | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Osaka-Kobe | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Palembang | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Rangoon | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Shenzen | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Taipei | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Thành-Pho-Ho-Chí-Minh (Ho Chi Minh City) | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Tianjin | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Tokyo | 2010 | 2010 | 2010 | 2010 | 2025 | 2020 | 2020 | 2020 | 2025 | 2025 | 2025 | 2025 |
| Vancouver | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Xiamen | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |
| Zhanjiang | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 | 2010 |

## 3.3 Conclusions on the Use of Acceptable Risk

The use of acceptable risk involves similar information as is needed for an ROA – i.e. the value at risk and expected shortfall. The information can be used to estimate the *current* decisions on when it is optimal to undertake an investment and at what level to undertake it. To be sure this decision can change but the calculations can be taken at regular intervals and decisions revisited. It does not avoid the case where a previous decision to invest has already been taken and new information indicates that it would not have been taken if that information had already been present. Such cases, however, cannot be avoided under any system of decision-making.

The advantage of acceptable risk used in the framework of this kind of sophisticated risk assessment is that it allows political inputs to be process to be made relatively easily. Policy-makers, in participation with other stake holders can discuss this question, looking into costs and benefits, within existing institutions and arrive at an informed decision that has democratic support.

The other advantage of this method is that it can be applied without too much difficulty. The tools for its application are available, as has been demonstrated in this paper. We therefore see more scope for this approach to be applied in decision-making to take account of future learning and quasi-options values.

# 4 References

Abadie, L.M., Chamorro, J.M., 2013. Investment in energy assets under uncertainty: Numerical methods in theory and practice, 1st ed. Springer, London.

Abadie, L.M., Galarraga, I., Rübbelke, D., 2014. Evaluation of two alternative carbon capture and storage technologies: A stochastic model. Environ. Model. Softw. 54, 182–195. doi:10.1016/j.envsoft.2014.01.002

Basque Government, 2007. Metodología para valorar los costes de los impactos del cambio climático en el País Vasco. El caso de Bilbao - Methodology to value the costs of climate change impacts in the Basque Country. The case of Bilbao. Basque Government, Bilbao.

Chambwera, M., Heal, G., Dubeux, C., Hallegatte, S., Leclerc, L., Markandya, A., McCarl, B.A., Mechler, R., Neumann, J.E., 2014. Economics of adaptation, in: Field, C.B., Barros, V.R., Dokken, D.J., Mach, K.J., Mastrandrea, M.D., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 945–977.

DEFRA, 2006. Flood Risks to People Phase 2 (No. R&D Technical Report FD2321). DEFRA & Environment Agency, Flood and Coastal Defence R&D Programme.

DEFRA, 2004. The appraisal of human related intangible impacts of flooding (No. R&D Technical Report FD 2005/TR). DEFRA & Environment Agency, Flood and Coastal Defence R&D Programme.

DEFRA, 2003. Flood Risks to People Phase 1 (No. R&D Technical Report FD2317). DEFRA & Environment Agency, Flood and Coastal Defence R&D Programme.

DETR, 1999. Flood and coastal defense project appraisal guidance FCD-PAG3 Economic appraisal. UK Department for Environmental, Transport and Regions, London.

Diputación Foral de Bizkaia, 2008. Varios ríos vizcainos alcanzaron su máximo nivel de caudal edesde hace 20 años (Área de Hidrología. Departamento de Medio Ambiente). Diputación Foral de Bizkaia, Bilbao.

Eustat, 2014. Socio-economic statistics. Basque Autonomous Community [WWW Document]. URL www.eustat.es (accessed 10.2.14).

Fernández Gómez, M., 1993. Inundaciones en el País Vasco. Rev. Obras Públicas 1, 26–35.

Groom, B., 2014. Discounting, in: Markandya, A., Galarraga, I., Sainz de Murieta, E. (Eds.), Routledge Handbook of the Economics of Climate Change Adaptation, Routledge International Handbooks. Routledge, Oxon (UK) and New York (USA), pp. 138–168.

Hallegatte, S., 2009. Strategies to adapt to an uncertain climate change. Glob. Environ. Change 19, 240–247.

Hazera, J., 1968. La region de Bilbao et son arriere-pays: Etude geomorphologique. Munibe 20, 1–328.

Hull, J., 2006. Options, futures, and other derivatives, 6th ed. ed. Pearson/Prentice Hall, Upper Saddle River, N.J.

Hunt, A., Watkiss, P., 2010. Climate change impacts and adaptation in cities: a review of the literature. Clim. Change 104, 13–49. doi:10.1007/s10584-010-9975-6

Ibisate, A., Ollero, A., Ormaetxea, O., 2000. Las inundaciones en la vertiente cantábrica del País Vasco en los últimos veinte años: principales eventos, consecuencias territoriales y sistemas de prevención. Ser. Geográfica 9, 177–186.

Jeuland, M., Whittington, D., 2013. Water Resources Planning under Climate Change: A "Real Options" Application to Investment Planning in the Blue Nile (Discussion Paper No. dp-13-05-efd). Resources For the Future.

Jiménez Cisneros, B.E., Oki, T., Arnell, N.W., Benito, G., Cogley, J.G., Döll, P., Jiang, T., Mwakalila, S.S., 2014. Freshwater resources, in: Field, C.B., Barros, V.R., Dokken, D.J., Mach, K.J., Mastrandrea, M.D., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 229–269.

Kontogianni, A., Tourkolias, C.H., Damigos, D., Skourtos, M., 2014. Assessing sea level rise costs and adaptation benefits under uncertainty in Greece. Environ. Sci. Policy 37, 61–78. doi:10.1016/j.envsci.2013.08.006

Kovats, R.S., Valentini, R., Bouwer, L.M., Georgopoulou, E., Jacob, D., Martin, E., Rounsevell, M., Soussana, J.-F., 2014. Europe, in: Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1267–1326.

Markandya, A., 2014. Incorporating climate change into adaptation programmes and project appraisal: Strategies for uncertainty, in: Markandya, A., Galarraga, I., Sainz de Murieta, E. (Eds.), Routledge Handbook of the Economics of Climate Change Adaptation, Routledge International Handbooks. Routledge, New York, pp. 97–119.

Mosegaard, K., Sambridge, M., 2002. Monte Carlo analysis of inverse problems. Inverse Probl. 18, R29.

Myers, S.C., 1977. Determinants of corporate borrowing. J. Financ. Econ. 5, 147–175. doi:10.1016/0304-405X(77)90015-0

Olcina, J., Sauri, D., Hernández, M., Ribas, A., 2016. Flood policy in Spain: a review for the period 1983-2013. Disaster Prev. Manag. Int. J. 25, 41–58. doi:10.1108/DPM-05-2015-0108

Osés Eraso, N., Foudi, S., Galarraga, I., 2012. Análisis del impacto socio económico del daño por inundación en la Ría de Nervión: un cambio de escenario ante la apertura del Canal de Deusto. Informe de Avance del Proyecto. BC3 Basque Centre for Climate Change, Bilbao.

Penning-Rowsell, E., Johnson, C., Tunstall, S., Tapsell, S., Morris, J., Chatterton, J., Green, C., 2006. The benefits of flood and coastal risk management. A manual of assessment techniques (Multi-coloured Manual). Flood Hazard Research Centre and Middlesex University.

Refsgaard, J.C., van der Sluijs, J.P., Højberg, A.L., Vanrolleghem, P.A., 2007. Uncertainty in the environmental modelling process – A framework and guidance. Environ. Model. Softw. 22, 1543–1556. doi:10.1016/j.envsoft.2007.02.004

Rockafellar, R.T., Uryasev, S., 2002. Conditional value-at-risk for general loss distributions. J. Bank. Finance 26, 1443–1471. doi:10.1016/S0378-4266(02)00271-6

Rodríguez, A., Martínez, E., Guenaga, G., 2001. Uneven Redevelopment New Urban Policies and Socio-Spatial Fragmentation in Metropolitan Bilbao. Eur. Urban Reg. Stud. 8, 161–178. doi:10.1177/096977640100800206

SAITEC, 2007. Estudio hidráulico de la apertura del Canal de Deusto (No. P0744-SR-EST-MEM001-V03). Bilbao City Council, Bilbao.

Taylor, T., 2006. Valuing cultural heritage and climate change risks, in: Quantifying the Cost of Impacts and Adaptation in the UK.

Watkiss, P., Hunt, A., 2013. Real Options Analysis: Decision Support Methods for Adaptation (No. Briefing Note 4), MEDIATION Project. Funded by the EC's 7FWP.

Watkiss, P., Hunt, A., Blyth, W., Dyszynski, J., 2015. The use of new economic decision support tools for adaptation assessment: A review of methods and applications, towards guidance on applicability. Clim. Change 132, 401–416. doi:10.1007/s10584-014-1250-9

Wilmott, P., 2014. Paul wilmott on quantitative finance. Wiley, Hoboken, N.J.

Woodward, M., Gouldby, B., Kapelan, Z., Khu, S.-T., Townend, I., 2011. Real Options in flood risk management decision making. J. Flood Risk Manag. 4, 339–349. doi:10.1111/j.1753-318X.2011.01119.x

Zeng, S., Zhang, S., 2011. Real Options Literature Review. iBusiness 03, 43–48. doi:10.4236/ib.2011.31007

# Sub-task 3: Methods for expressing risk and ambiguity in economic analysis

# Executive Summary

This paper reviews methods used by economists to capture the influence of social preferences on risk in economic appraisal. It begins with a graphical illustration of how different risk preferences in a gambling example can yield differing utility functions. Risk aversion parameters are introduced as measures of a society's preference for certainty in project alternatives. A brief survey of risk parameters in climate change economic literature shows that risk aversion parameters vary from 1 to 4.

Section 2 includes a wide review of literature on tools for factoring risk preferences into calculations of the social discount rate used in project appraisal. A simple model including a Ramsey term capturing risk as variability in growth rates is presented as the first basic model. Traeger's intertemporal risk coefficient, shown to have a greater impact on discount rates than the Ramsey term, is presented and discussed, as well as an alternative Relative Ambiguity Aversion factor that includes a time-based growth factor. A Relative Risk Aversion model is introduced and dismissed in favour of the previously discussed intertemporal model.

In the literature review, a model of inequality aversion proposed by Gollier (2001) is also reviewed. Based on the marginal utility of income, this wealth premia model assumes a concavity of risk aversion over wealth and suggests that a linear interest rate does not capture these shifts in preferences across income. An example experiment conducted amongst Swedish university students is presented as an illustration of this inequality consideration, with a model for a Social Marginal Rate of Substitution discussed. A brief selection of cases around the world show that risk preferences can be derived from market behaviour in insurance products, individual surveys and interviews and technology adoption decisions.

Section 3 explores the application of risk preferences in climate-relevant economic appraisal work. Recommendations from a US EPA-commissioned panel supports the use of declining discount rates (DDR) and expected net present value (ENPV) approaches to projects where uncertainty is a major concern. DDR is preferred for managing uncertainty over the economy, while ENPV can normalise differing opinions amongst economists. Cases in Brazil and India are presented to illustrate specific considerations in measuring risk preferences, namely the presence of a concave risk preference function and a role for country-specific risk aversion models that factor in social structures in particular countries.

In order to inform the common practice of transferring values on preferences between sites, Section 4 discusses latest developments on benefits transfer in cost benefit analysis with consideration toward how risk preferences may play into these practices. Different strategies for benefit transfer include unit transfer, income-adjusted unit transfer and value function transfer. The latter is recommended by practitioners for its ability to capture site-specific differences.

Other best practices include the use of meta-analyses for data, consistency in method used to collect data on preferences and controlling for differences across a range of social, economic and natural science factors. A decision rule from Barton (2006) considers the risks of transferring preferences from one site to another, offering a model with population affected by an appraisal and estimates of error in calculations compared with the cost of an additional preferences study in the transfer site. Transfer error is also discussed along with strategies to reduce different types of error in benefits transfers. Issues of double counting, sectoral preferences, altruism, and the decision context are all reviewed.

Section 5 considers the implications of risk preferences literature and cost benefit analysis best practices to recommend specific options for use in economic appraisal. Intertemporal risk aversion is identified as the most robust tool for reflecting risk preferences in a social discount rate. This method has yielded similar results to the use of a wealth premia factor. Declining discount rates or an ENPV approach should be employed depending on the characteristics of uncertainty facing a particular appraisal. The paper ends with a note of caution on the availability of data for applying any of the methods discussed above.

# 1 Introduction

Advanced economic appraisal attempts to provide a robust picture of costs and benefits for the life of a project or programme. Climate adaptation projects are likely to have prolonged lifetimes for benefits from avoided damages from climate change, potentially spanning multiple generations. Social preferences around time and risk help inform trade-offs between present consumption and investment in future welfare changes. This paper examines the latest developments in considering risk preferences amongst populations when conducting economic appraisals, reviews best practices for incorporating these preferences into cost-benefit analyses and suggests which of the alternatives discussed may prove useful to analysts and decision makers.

## 1.1 Risk and decision making

The role of risk coefficients in economic appraisal is to parameterise the attitude to risk that a decision-maker has. Preferences for risk typically are described in terms of the decision-maker's attitude toward actuarially fair gambles. An actuarially fair gamble exists when the mean outcome of the gamble is equal to the 'price' of playing. Suppose, for example, an individual is offered the privilege (free of charge) of playing the following gamble: They *receive* €1 if a tossed coin lands on 'heads'. However, they must *pay* €1 if the coin lands on 'tails'. Since the probability of 'heads' or 'tails' is one half, this is an actuarially fair gamble. The expected value of the gamble is

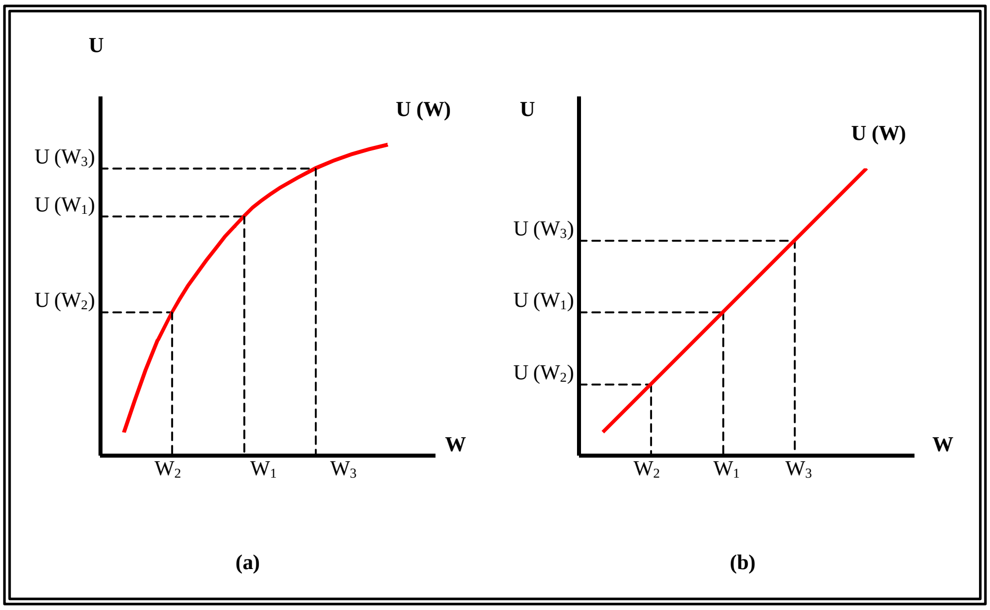
$0.5 \times €1 + 0.5 \times -€1 = €0$

which is the 'price' paid to play.

If the decision-maker rejects all such gambles, then they are said to be risk-averse. The risk-averse decision-maker may also be viewed as someone who is willing to pay a positive amount to avoid risk. If the decision-maker prefers to take actuarially fair gambles, then they are said to be a risk-lover; such an individual would pay for the privilege of participating in the gamble. A decision-maker who is indifferent to such gambles is said to be risk-neutral.

These different preferences to risk are captured in the decision-maker's utility function. The figure below displays the utility function for a risk-averse (panel a) and risk-neutral (panel b) decision-maker. Utility is measured on the vertical axis and outcomes on the horizontal axis. Assume the risk-averse decision-maker in panel (a) starts with $W_1$ (baseline 'wealth'). The corresponding level of well-being or utility is $U(W_1)$. Now, suppose (s)he is offered a bet of €5 on the toss of a coin, which is accepted. If they lose, the new level of wealth will be $W_2$, where $W_2$ equals $W_1$ minus €5. Alternatively, if they win, the new level of wealth will be $W_3$, where $W_3$ equals $W_1$ plus €5. However, since this decision-maker is risk-averse, they are not concerned with additions to wealth per se. Rather, they are interested in changes in utility. One can see from panel (a) that the absolute magnitude of the loss of utility associated with losing the gamble, $U(W_1)$ minus $U(W_2)$, is greater than the gain in utility from winning, $U(W_1)$ plus $U(W_3)$. In contrast, for the risk-neutral decision-maker, the absolute magnitude of the changes in utility are equal for the €5 loss or the €5 gain – that is, $U(W_1)$ minus $U(W_2)$ equals $U(W_1)$ plus $U(W_3)$.

**Figure 1:** Utility Functions and Risk Preferences



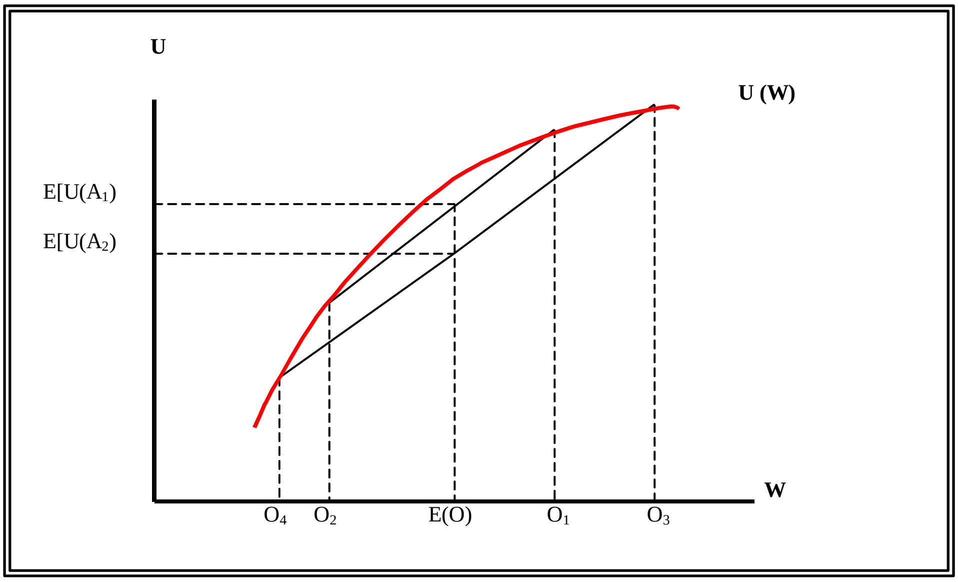

How can this method be used to make comparisons between options under conditions of risk? To answer this question consider Figure 2 below. Suppose that adaptation option $A_1$ yields a NPV of $O_1$ or $O_2$, depending on which state-of-nature occurs, with probabilities $p$ and $(1 - p)$, respectively. Hence the expected NPV, ENPV(O), is given as $p \times O_1 + (1 - p) \times O_2$. In contrast, option $A_2$ provides a chance of a much greater NPV, $O_3$, but also a chance of a much smaller NPV, $O_4$. Given the risk-averse utility function shown, the expected utility of option $A_1$ is greater than that of option $A_2$ – i.e. $E[U(A_1)] > E[U(A_2)]$.[6] As more utility is assumed to be preferable to less utility, the decision-maker will select option $A_1$.

The parameter η characterizes the percentage decrease in marginal utility from a percentage increase of consumption and expresses aversion to fluctuations over time and with respect to risk. The larger is η, the greater is the curvature of U(W). One can also see from Figure 2 that the dispersion of outcomes associated with option $A_2$ is greater than for option $A_1$. A risk-averse decision-maker, when faced with a choice between two options with the same ENPV, will select the option with the smaller distribution of outcomes. Equally, if two options have the same dispersion of outcomes, but different ENPVs, the highest ENPV will provide the highest expected utility. However, if $ENPV(A_1) < ENPV(A_2)$ and $SD(A_1) < SD(A_2)$ the decision-maker must trade-off expected value with the level of associated risk. In order to select the 'best' option, the

[6] $E[U(A_1)] = p \times U(O_1) + (1 - p) \times U(O_2)$ and $E[U(A_2)] = p \times U(O_3) + (1 - p) \times U(O_4)$.

decision-maker must therefore estimate the expected utility associated with each option, using the method described above.

**Figure 2:** Option Selection Using Expected Utility: Risk-averse Decision-maker




In the case where there are alternative probability sets, decision makers may have aversion to the ambiguity they face in choosing between these sets. Ambiguity aversion leads the decision-maker to select the adaptation option that is the highest ranked when the lowest expected utility of each is considered. In other words, which option has the highest lowest expected utility?

The empirical questions for this sub-task are: **how can the risk preferences of decision makers be included in economic analysis of adaptation**? And: **what values can be used in the adaptation context**?

The values estimates found in an initial survey of the literature are limited in number. The values discussed in discussion of climate change mitigation include: Dasgupta (2008): 2-4; Weitzman (2007): 1-4; Stern (1976): 2; Stern (2006): 1.

**Table 1.** Risk aversion parameter estimates expressed in economic literature

| Study | RA parameter estimate |
|---|---|
| Mirrlees (1971) | 2 |
| Stern (1976) | 2 |
| Stern (2006) | 1 |
| Weitzman (2008) | 1-4 |
| Dasgupta (2008) | 2-4 |

# 2 Evaluation of existing literature

Literature on the treatment of risk in cost-benefit analysis calculations concerns the question of how to transfer values from one project or programme to evaluate similar proposals without undergoing costly or unavailable onsite analysis (Gollier, 2001).

Differences in social discount rates have defined debate over integrated assessment models for climate change and mitigation policies (Traeger, 2009). Risk aversion is incorporated into the Ramsey (1928) social discounting rate equation with the following expanded formula:

$$r = \delta + \eta u - \eta^2 \frac{\sigma^2}{2} \tag{1}$$

where 'σ' represents standard deviation of the growth rate 'μ'. As 'σ' is expressed in terms of percentage and 'η' as a full unit, the third term in this equation is often very small, minimising the impact of risk as it relates to volatility among possible outcomes on the discount rate. Risk is still expressed in part in the second term of the formula, as 'η' expresses a preference for consumption smoothing over time and population. This model requires the assumption that consumption smoothing over time fully captures risk aversion preferences in a population. More direct methods of accounting for risk in social discounting are explored below.

## 2.1 Accounting for intertemporal risk

Traeger (2009) proposed an additional term to the Ramsay equation to account for intemporal risk aversion. The coefficient 'RIRA' is expressed as a function of intemporal elasticity of substitution ('η') and risk aversion in terms of volatility of outcomes ('σ') in the following form:

$$r = \delta + \eta u - \eta^2 \frac{\sigma^2}{2} - RIRA|1 - \eta^2| \frac{\sigma^2}{2} \tag{2}$$

This separate notation aims to disentangle risk aversion preferences from consumption smoothing preferences, represented by 'η' in the above equation. 'RIRA' is estimated to have a much larger effect on *r* than the third term in the equation. Traeger estimates that RIRA can be as high as 26.5, showing a substantial effect on the output of the equation. With a high degree of uncertainty, as is often presented by climate change, this risk factor could cancel out the impacts of projected growth on a social discount rate, which has important implications for policymaking activity.

## 2.2 Uncertainty over time

An alternative to modelling intemporal risk aversion, uncertainty over time can be directly controlled for in the Ramsey equation. This approach has the benefit of accounting for changes in risk as time frames expand. Traeger introduces a Relative Ambiguity Aversion coefficient that measures reaction to uncertainty, building on literature showing a dislike of ambiguous outcomes from choices and unknown probabilities. The equation

$$r = \delta + \eta u - \eta^2 \frac{\sigma^2 + \tau^2}{2} - RAA|1 - \eta^2| \frac{\tau^2}{2} \tag{3}$$

where 'τ' represents time period, provides more flexibility in the social discount rate as a time period grows.

Gierlinger and Gollier (2009) show that ambiguity aversion can indeed decrease the social discount rate. However, their work discusses two diametric ways ambiguity can affect the social discount rate. First, an ambiguity prudence effect reduces the discount rate with increased uncertainty similar to 'σ' growth uncertainty above. The authors find, though, that ambiguity-neutral discount rates are often higher than socially-efficient rates that do not account for ambiguity aversion.

## 2.3 Relative Risk Aversion in Ramsey discounting

A more general measure of risk aversion, Relative Risk Aversion (RRA), reduces preferences for a particular path as uncertainty around that option grows. Values for the RRA parameter in resource consumption vary between 0 and 5. Howitt et. al. (2005) found an RRA of 1.5 for water-level management in California.

Applied to climate change, RRA appears to be directly correlated to emissions taxes, showing that uncertainty reduces discounting (Ha-Duong & Treich, 2004). Often models underestimate the effect of RRA by not controlling for the effect of aversion to intertemporal substitution, which can dampen impacts from RRA. Other applications find that RRA has no significant impact on output (Howitt, Msangi, Reynaud, & Knapp, 2005).

Under the assumption that an agent only prefers early resolution of uncertainty to late resolution of uncertainty if there is a chance that he or she will receive information that will affect future decisions from such resolution, Traeger shows that, where intertemporal risk aversion is present in the Ramsey equation for social discount rates, the pure rate of time preference ('δ') must be zero. As a decision maker has no preference for uncertainty resolution except in the case of information availability, there must be no time impatience between near and far periods. This assumption stems from a theory of social welfare maximisation in which social planners do not differentiate between utility in the present period and utility at a later time, given equal utility from the value of a project in both time periods.

Intemporal risk aversion captures impatience in time that is attributed to an increase in uncertainty as the time horizon increases from a decision maker's perspective. This condition lends support to including a measure of intemporal risk aversion (RIRA as presented by Traeger) in any modelling of the social discount rate. Reliance on RIRA will prefer distant outcomes with stronger certainty, at times over better outcomes that are less certain. The use of Relative Risk Aversion parameters requires some knowledge of probability around possible outcomes. Where the likelihood of these outcomes is unknown, deep uncertainty must be included in models.

## 2.4 Alternative approaches to RIRA coefficient

Weitzman (2009) proposes a model with high uncertainty (fat tails) over time that requires an infinite willingness to transfer consumption into the future. His study proposes limiting this transfer value to the value of society's collecting statistical life.

Traeger (2009) analyses both risk aversion and intertemporal substitutability. He shows that increased risk aversion and decreased intertemporal substitutability lead to similar treatments of natural resources. In the case of irrigation water supply, reduced substitution leads to reduced aquifer pumping in good times and increased pumping in bad times.

Traeger criticises the standard model of risk aversion in a dynamic setting on two grounds. First, the model assumes that risk aversion coincides with consumption changes over time, requiring intertemporal risk neutrality. Though risk and time preferences are related—uncertainty is likely to increase over future timeframes, experimental results indicate that social preferences over various levels of risk and time follow different trajectories (Andreoni & Sprenger, 2012). Second, the model does not allow for aversion to second-order uncertainty. Resource economic theory holds that proper accounting for these attitudes have significant implications for extraction of fossil fuels and other finite resources. These models show that the income effect of increased risk aversion or future risk reduces use of resources in the present, while a substitution effect between consumption now and in the future reduces this impact. Where an aversion to intertemporal substitution is present, the income effect dominates. Risk aversion is also found to lower the certainty-equivalent social discount rate.

## 2.5 Wealth Inequality and Risk

Gollier (2001) shows that wealth inequality affects the equilibrium risk-free rate of return on investments, specifically that this rate is reduced by wealth inequality if the inverse of absolute prudence is concave in wealth. If inequality is not factored into models of consumption, the risk-free rate may be overpredicted.

The paper first disputes the claim that risk aversion is constant with wealth. Weil (1992) shows that wealth inequality reduces price of equity if an actor is more risk averse than the average equity-holder. As risk averseness decreases with wealth, poor people are more risk-averse while rich people are less. This is an extension of the concept of diminishing marginal utility of income—as basic needs are met, individuals are less dependent on additional income and derive less utility from each additional unit.

Gollier shows that the only condition where inequality-adjusted risk preferences do not differ from standard risk preferences is when harmonic absolute risk aversion (HARA) exists. A similar principle applies for situations where absolute risk aversion is linear. For those functions where HARA does not apply, an equity premium must be identified.

A principle assumption of Gollier's theory is that risk tolerance has a concave distribution across wealth—that is, risk tolerance grows with wealth until a certain level (e.g. wealth levels at retirement) where it levels off. He presents evidence for a weak, but affirmative case to support this assumption. While the evidence is weaker for the purchase of equities versus bonds among individuals, concavity of risk aversion should exist for cities and countries facing climate change risk. While wealthy communities such as London have taken steps to protect significant wealth (e.g. Thames flood barrier), risk aversion is likely to be concave over wealth due to high fixed costs of climate adaptation. Less wealthy communities are still investing in adaptation measures (though not necessarily with their own capital), which have high fixed costs. As risk aversion concavity can be assumed for climate adaptation, the wealth/equity premium should be accounted for in cost benefit analyses. As we will see below, this can be accomplished through the inclusion of a risk aversion term in the Ramsey formula.

Gollier's second finding with respect to wealth inequality builds on the assumption of concavity of risk aversion. He shows that the risk-free rate of interest is reduced if absolute imprudence, or a disregard towards savings against future growth uncertainty, is distributed concavely across wealth. Similar logic applies for imprudence as does for risk aversion. More prudent individuals are likely to save a greater share of income in the face of future uncertainty. In the case of prudence, lower risk-free rates are required to encourage consumption over saving, leading to a negative relationship between prudence and risk-free rates. As wealth and risk-seeking increase, imprudence can be expected to follow a similar path.

Taken from an empirical perspective, Gollier's analysis shows that risk-free rates can be overpredicted when wealth inequality is not considered in models. While his analysis does not offer a direct method of applying an equity premium to the Ramsey formula, this bias should be considered when calculating discount rates for climate adaptation projects. For example, models predicted a constant four percent risk-free rate, but the average over the past century has been closer to one percent (Gollier, 2001). The wealth premium can have a similar effect on the discount rate as the intemporal risk aversion rate introduced by Traeger. As the impact of a wealth premium is conditioned on risk aversion concavity, these two qualifiers should not be used in combination, but rather as two alternative methods of controlling for risk preferences in a discount rate. In both cases, the risk preference or wealth premium term reduces the discount rate.

## 2.6 Estimating risk preferences

An experiment with 324 Swedish university students revealed that risk aversion was a natural preference in 91% of respondents, with 45% of respondents revealing a risk aversion coefficient between 2 and 5 (Carlsson, Daruvala, & Johansson-Stenman, 2005). This experiment equated inequality and risk aversion preferences in a survey between lottery choices. Risk aversion coefficients between 2 and 5 are equated to inequality aversion parameters between 0.22 and 0.56, though 31% of respondents reported these scores compared to 45% reporting equivalent risk aversion preferences.

This study found a social marginal rate of substitution (SMRS) between compensation to high and lower-income populations based on inequality aversion. The authors calculate the SMRS with the following formula:

$$SMRS_{ij} = \frac{y_i^{-\rho}\Phi^{-\gamma(1-\rho)-\gamma\frac{\bar{u}(y_i-\bar{y}-\sigma\Phi)(1-\rho)}{\bar{y}\sigma\Phi}}}{y_j^{-\rho}\Phi^{-\gamma(1-\rho)-\gamma\frac{\bar{u}(y_i-\bar{y}-\sigma\Phi)(1-\rho)}{\bar{y}\sigma\Phi}}} \tag{4}$$

where ‘γ’ represents inequality aversion and ‘ρ’ risk aversion. In this situation, an inequality aversion parameter of 0.2 (near the median of the sample response), would yield equal utility from proportional distribution policies. For example, taking 100SEK from a person with an income of 20,000SEK and giving 50SEK to a person with 10,000SEK would be seen as equivalent in terms of social welfare created.

These results first indicate that risk aversion may be a stronger preference than inequality aversion, and that the former can be inclusive of the latter in a discounting calculation. Second, an SMRS calculation can be used to model trade offs between income groups in a country and the impact of these trade offs on social welfare.

Other studies that aim to develop national measures of risk may be of interest to the discussion in this paper. Attempts to evoke risk preferences from representative surveys can highlight differences between countries, which may have bearing in calculating place-based risk-contingent discount rates for particular projects and programmes. An attempt to compare U.S. and Chinese risk preferences found that Chinese respondents had a greater appetite for risk-seeking behaviour (Hsee & Weber, 1999). Scholarship constructing risk aversion models from insurance products attempt to provide an accurate picture of risk preferences based on actual market decisions, but these aggregations encounter a high level of heterogeneity in national markets and overemphasise small changes (Cohen & Einav, 2007; Snydor, 2010). In the public sector, risk preferences over alternative strategies for Australian forestry policy were derived from in-depth interviews with stakeholders (Ananda & Herath, 2005). Considering applications for climate adaptation projects, higher risk aversion has been shown to create an adoption lag for agricultural technology in Chinese cotton farmers (Liu, 2013).

# 3 Risk premia in practice

A 2012 study commissioned by the U.S. Environmental Protection Agency suggested that 1) the Ramsey formula is an adequate framework to evaluate intemporal valuation questions under uncertainty, 2) theory supports the use of declining discount rates over time and controlling for certainty and 3) that the uncertainty of projected growth over time can explain the use of declining discount rates over time but that dynamics in other factors of the Ramsey formula are less clear (Arrow et al., 2012). In a review of approaches among resource economists, the EPA panel identifies two main branches of discounting over long time horizons: declining discount rates (DDR) and expected net present value (EPNV). The former calls for a stepwise decrease in discount rates applied to projects as time horizons for realised benefits grow (similar to the approach the United Kingdom's government has taken). The latter averages different growth scenarios to yield a mid-range discount rate. The panel argues that evaluators apply an EPNV approach when disagreement among experts on Ramsey parameters exists and a DDR approach when uncertainty exists over the state of the future economy. Both have implications for valuing projects with uncertainty and long-term impacts, which climate adaptation projects often include. The source of the uncertainty can help inform which approach is more appropriate for each situation.

A debate has emerged between Weitzman and Gollier on how to account for uncertainty over distant time horizons has been followed closely in environmental economics. Weitzman's argument (1998) that ENPV should be employed to represent a probability-based level of discounting yields a diminishing discount rate, bolstering the welfare of future generations when compared to other approaches. He presents the ENPV approach as an insurance policy against catastrophic risks (2009). Gollier shows that Net Future Value over distant horizons increases rather than decreases and resources would create more welfare if they were saved in the present for future use (2002). Freeman (2010) argues for Weitzman's approach, citing a general preference for smoothing consumption over time. A bias towards caution (lower discount rates for long horizons) may be prudent given the exclusion of deep uncertainty events such as climate feedbacks that could yield higher damages than expected by existing models.

While literature outlines alternative approaches to accounting for risk in discounting, both concepts discussed above require empirical data to determine the impact of risk preferences on discount rates. We will now review findings of risk preference terms in empirical data across different geographic areas.

Issler and Piqueira (2000) study the Brazilian economy to measure the impact of risk aversion and wealth premia on discount rates. While the relative risk aversion coefficient is high, the overall discount rate remains high due to strong growth projections. They find a risk aversion coefficient between 1.10 and 4.89, indicating the potential for relatively high risk aversion among Brazilians, though high variability in these findings make it difficult to compare to other economies. Across all measures though, Brazilians display two to three times the level of risk aversion when compared to U.S. consumers. The authors do not find strong evidence of an equity premium in Brazil, indicating little impact on discounting from wealth inequality. This is surprising as Brazil has a relatively high, though decreasing, Gini coefficient of inequality (The Economist Online, 2011). These findings lend support to the theory discussed above that risk aversion increases concavely with wealth, at least between countries, if not within.

Further, different countries may be structured socially such that risk preferences do not transfer equally between countries. Mazzocco & Saini (2007) show that the caste system in India creates a separate level of social decision making, where risk-sharing takes place at the caste level rather than the market or geographic level. The authors show that previous models of risk-sharing assume identical risk preferences between households, which is an unrealistic assumption. They propose a test of whether risk preferences in a study area are homogenous that compares expenditure functions of two test households, scaled up to an economy level. Their findings suggest that risk preferences can differ from country to country and an econometric test may be able to reveal whether values in one country will transfer to another.

These cases demonstrate that discount rates and treatment of risk cannot be easily transferred from one location to another. Rather, idiosyncratic information for each country's economy will determine which discount rates and risk preferences should be applied to a project or programme. Econometric methods can help to identify which economies are suitable candidates for value transfers.

# 4 Transferring value between locations

In addition to discussion of methods to embody risk measurements in quantitative expressions of economic preferences, this paper aims to provide guidance for transferring calculations of costs and benefits of a climate adaptation project from one situation to other similar situations. This process, known as benefits transfer, is formally defined as follows:

> **Benefits transfer:** the transfer of existing estimates of non-market valuation to a new study which is different from the study for which the values were estimated

When estimates are transferred from one study site to a policy site, inherent assumptions and biases are included. While using benefits transfer to assess an impact of a project or programme can save much time and resources (or may be a necessity where data is not available), it is important to guarantee that such an approach will yield valid results. Understanding details and preferences between sites can inform whether a benefits transfer approach is appropriate. In a 2006 review of the issues related to benefits transfer, a report for the Organisation for Economic Cooperation and Development (OECD) identified inaccuracy in benefits transfer as inevitable, but suggests that some level of inaccuracy may be acceptable for a valid transfer (OECD, 2006). The report cites baseline environmental quality data as an essential background input to collect before applying benefits transfer from another site.

Benefits transfer can be conducted by either transferring units (i.e., the value of a statistical life or the willingness to pay for ecological services from a hectacre of forest) from one location to another or by transferring functions from site to site. The first method—a 'unit value transfer' approach—can be adjusted by average income (i.e., if the policy site has an average income equivalent to 45% of the policy site's income, values can be reduced to that level, given the income elasticity of the valued good or service). Overall, the unit transfer approach requires suspension of the possibility that preferences in two different sites may vary. Direct transfers neglect site-specific information such as: socio-economic and demographic characteristics, physical and environmental attributes of a site and differences in 'market conditions' (such as availability of substitutes). When adjusting for income, enough data must be available to determine income elasticity for the good being valued. Navrud and Ready (2006) raise the question of how much additional uncertainty is introduced by using benefits transfer methods.

The second approach to benefits transfer mentioned above—value function transfer—converts a relationship for the same good in two different sites by transferring a value function responsive to specific characteristics at each site. Navrud and Ready propose the following value function for calculating willingness to pay:

$$WTP_{ij} = f\left(G_j, H_i\right) \tag{5}$$

where $G_j$ are the characteristics of the environmental good of interest at the policy site and $H_i$ are the characteristics of households at the study site. The function attempts to delineate the value for environmental goods given specific household characteristics and calculate this value for a different set of households. To avoid statistical errors from transferring values from a single site, practitioners often rely on meta-analyses to yield significant results for *G* values in the value function. Mourato and Atkinson suggest the following stylised function in the OECD report on benefits transfer (2006):

$$WTP_i = a_1 + a_2 TYPE\ OF\ SITE + a_3 SIZE\ OF\ CHANGE + a_4 VISITOR\ NUMBERS + a_5 NON\ USERS + a_6 INCOME + a_7 ELICITATION\ FORMAT + a_8 YEAR \tag{6}$$

## 4.1 Using meta-analysis for benefits transfers

When using values from meta-analyses, Navrud and Ready suggest selecting studies that utilise similar methodologies. Bergstrom and Taylor (2006) caution against the widespread application of meta-analysis approaches to benefits transfer, calling for improved methodological checks for this approach. In order to strengthen meta-analysis benefits transfer, they stress the importance of 'commodity consistency' of environment goods included in studies that form the meta-analysis. This can be proxied by substituting the services provided by an environmental good for the good itself. Limiting meta-analyses to sites that provide similar services from environmental goods improves the validity of the values for transfer, but may reduce the pool of available data for a desired benefits transfer. If commodity consistency cannot be achieved, heterogeneity between commodities must be controlled for in the value function (i.e., if various-sized wetlands are included in a meta-analysis, the size of wetland can be included as a controlling factor in a value function).

Additionally, Bergstrom and Taylor argue that any meta-analysis must satisfy 'welfare change measure consistency.' For this condition to hold, the methods of estimating value for a given service from an environmental good must be equivalent. Either the same method (i.e., travel cost method, contingent value method, hedonic pricing, etc.) must be applied in each study, or an adjustment reflecting the theoretical differences in valuation methods must be included in calculations for the meta-analysis benefits transfer.

Use of benefits transfer has increased over time, especially in publicly-funded projects. The governments of Canada, the United States and the United Kingdom have developed a joint database of primary valuation studies. This database (found at www.evri.ca) compiles original studies for use in benefits transfer applications around the world. While database approaches to calculating benefits transfers increase the ease and applicability of benefits transfer in project valuation, experts caution against applying values from other studies without careful analysis of any necessary adjustments required for the policy site (OECD, 2006).

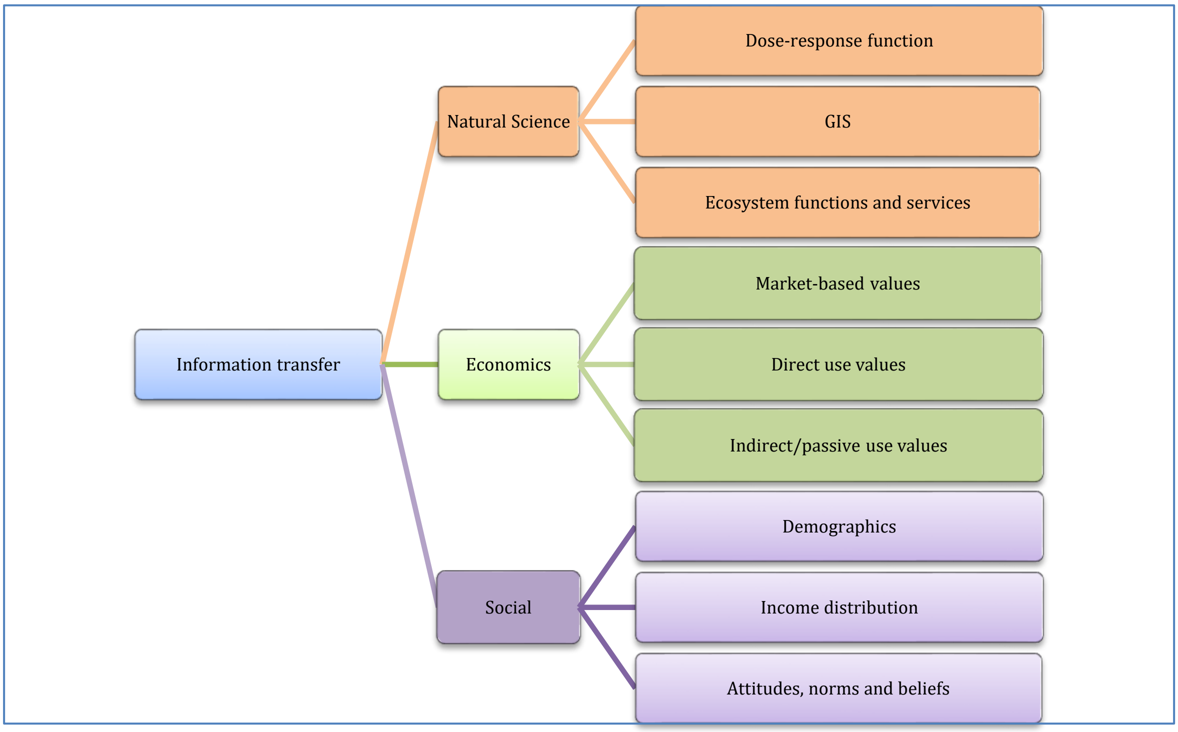
Brookshire (1992) notes that the level of accuracy required from a benefits transfer depends on the intended use of the data, with compensatory damages requiring the highest confidence in values, followed closely by policy decisions. Less binding uses, such as early screening or scoping of policy alternatives may remain useful even with high transfer errors.

Transferring values between countries has been common practice by international organisations such as the World Banks (Silva and Pagiola, 2003). Ready et al. (2004) examines the issues of whether intercountry benefits transfers create specific problems for the validity of cost benefit analysis in these cases. Controlling for individual characteristics, this study measured willingness to pay values for sickness in five European countries, finding significant disparity among the values placed on health in Spain and Portugal (with high values) and England (low values). This suggests some differences in preferences between countries that may otherwise appear similar in characteristics.

Wilson and Hoehn (2006) review modern issues in benefits transfer and show that, as a method for economic valuation, benefits transfer has grown increasingly influential in the past three decades. As use of the benefits transfer method has increased, so has attention to reducing errors accompanying this method. Suggestions for improving the validity of benefits transfer include the need to account for differences between population income and biophysical context in respective sites.

Spash and Vatn (2006) illustrate the various dimensions that must be accounted for in any value transfer. Figure 3 demonstrates the various factors in a value function that can differ between sites. When comparing two sites, differences between natural science and social groupings shown in Figure 3 should be accounted for in a value function. Parallels in economic preferences can be established through previously published studies, if available, but valuation methods should be conducted with care as to reduce any error in the study site, as discussed below. Site-specific risk preferences may be adjusted for under 'Attitudes, norms and beliefs' in this framework.

**Figure 3.** Site-specific factors in benefits transfer.




*Source: (Spash and Vatn, 2006)*

Barton (2006) discusses the trade-off between collecting original data at a policy site and uncertainty caused by transferring benefits information from other sites. In a stylised example, the analysis shows that the level of acceptable risk is an important determinant in choosing the welfare-maximising method to calculate costs and benefits for a project. By calculating data collection costs and costs of risk from an invalid estimation, project analysts can measure whether benefits for a project should be based on a previous study or if resources should be applied to collecting some level of on-the-ground data at the policy site to reduce uncertainty in the economic valuations of the project. This decision rule is modelled as follows:

$$Min\ E\ (W) = \ POP\ |\mu - c|\ \phi\ \left(\frac{0-|\mu-c|}{\sigma_{\mu-c}}\right) + gRC(n)$$

(6)

In this model, the following variables are measured:

POP= number of households affected by the policy under consideration

$|\mu\text{-}c|$ = the absolute value of net benefits per month resulting from the policy

$\Phi$ = the probability of net costs from an incorrect decision resulting from benefits transfer

$\sigma_{\mu\text{-}c}$ = standard error of net benefits

$g$ = price of an increase in information effort (data collection)

$R = r/(1-(1+r)^{-T})$, with r = the discount rate and T = time periods

C(n) = total information effort spent on the $n^{th}$ additional study

The model shows that the cost of performing a study is compared with the risk around anticipated benefits transferred from another study site. It can be used to inform different confidence levels set by the policy maker (i.e., if a 90% confidence level is acceptable, benefits transfer may be used where an onsite study would be necessary for a 95% confidence level). Efforts to reduce double-counting of risk are discussed below.

## 4.2 Transfer error in benefits transfer

The 'transfer error' is a value calculated by comparing benefits transfer from a study site to real data measured in a policy site. This measure can be calculated, where data is available, with the following formula:

$$Transfer\ Error = \frac{(transferred\ estimate - own\ study\ estimate)}{own\ study\ estimate} * 100 \quad (7)$$

Challenges related to value transfer include: a paucity of quality existing studies to transfer from, a mismatch in a change in a good from a proposed policy and the observed change in an existing study, differences in study sites and policy sites unaccounted for in the transfer formula, an incomplete or overgenerous determination of the market at a policy site and potential bias from aggregating individual components of value for a good. Use values—preferences measured by real expenditures such as fee payments, travel costs, lost wages, etc.—are likely to be more accurate than non-use values, which can suffer from a hypothetical bias.

Rosenberger and Stanley (2006) define three sources of transfer error present in benefits transfer applications: measurement, generalisation and publication, as outlined in Figure 4. In the first instance, measurement errors can occur from the inclusion of biases in collecting primary information for an initial test site. Measurement error can result from the valuation method used, flawed samples or any other problems with initial collection methodology. Second, generalisation errors result from applying values from one site to another without appropriately correcting for differences between sites. Finally, a publication selection bias results from careful selection of which studies are published, excluding those that may not support theory or that lack statistical significance.

**Figure 4.** Types of error in value transfers

| Transfer Error Elements | | | |
|---|---|---|---|
| **Type** | Measurement Error | Generalisation Error | Publication Selection Bias |
| Includes | Valuation errors, sampling errors, excluded populations | Differences from income, preferences, types of environmental goods | Favouring studies that support existing theory, only publishing statistically significant findings |
| Strategies for reduction | Careful data collection, uniform preference measurement methods (Contingent Valuation, hedonic pricing, etc.) | Adjust for differences between sites, choose similar sites and environmental services | Pull studies from open-access/grey literature inventories such as the Environmental Valuation Resource Inventory |

In the value for health study mentioned above, Ready et al. (2004) find a transfer error of 37-39% in transferring values between European countries. The size of this error is minimally affected by adjusting for income differences in countries or employing a value function transfer approach. Reviews of transfer errors in environmental studies find a wide range of errors from transferring willingness to pay estimates across countries (Brouwer, 2000; Rosenberger & Loomis, 2003). Bergland, Magnussen and Navrud (1995) also find transfer errors of 18-45% for both unit transfers and value function transfers in a study of non-monetary values for lakes in Norway.

Navrud and Ready (2006) idenitify the primary lessons learned over 30 years of conducting value tranfers. They caution that goods at a study site must be similar to the policy site values are transferred from in the definition of the good, the population affected and the degree of change in the stock of the good by a given policy. To combat omitted variable bias, they advise that maximum information be controlled for in a value transfer function. Best practices for reducing transfer errors include defining a level of acceptable error before calculating variance between transferred and real values and performing preference calibrations between sites. While additional research will continue to improve accuracy and understanding around benefits transfer, some level of transfer error must be accepted for current policy analysis. Ultimately, transfer errors represent the acceptable level of risk a decision maker is willing to incur to avoid the costs of conducting full studies at policy sites.

## 4.3 Double-counting risk in project analysis

The risk of double-counting and importance of accuracy in valuation of environmental problems has been a key debate since the publication of Costanza's global ecosystem services valuation in 1997, which estimated the value of the entire ecosystem to be as high as $61 trillion. Literature on double-counting in the context of ecosystem services valuation discusses overlap in definitions, conflict between valuation methods and lack of understanding of ecosystem complexity as the primary sources of double-counting in environmental valuation (Fu et al, 2010). Because ecosystem services are not sold on the market, accurate prices are not available and various attempts to unveil market prices may cause double-counting if not conducted using similar methods.

Outside of potential double-counting errors in initial valuation, double-counting can manifest in accounting for risk in a discount rate as well as deciding whether to use benefits transfer. This can be avoided in both cases. In the first instance, discount rates accounting for risk and uncertainty should use either expected net present value (ENPV) approach set forward by Weitzman or include a measure of risk preference within the Ramsay equation for a discount rate, but not both. On the same note, ENPV calculations should be between values that have not already been adjusted to reflect risk preferences of a population. In the second case, where risk-adjusted discount rates are used in the Barton decision model, analysts should be clear that these calculations have been made so that risk adjustments are not repeated in interpretations of the model.

## 4.4 Decision context

Decision context plays a role in determining risk preferences by shaping the background in which individuals make decisions. An important consideration in analysing risk preferences is the initial status of decision-makers. Kahneman and Tversky (1976) presented a ‘prospect theory’ that seeks to describe how initial endowments can explain decision outcomes more than a purely rational expected-utility model. Prospect theory suggests that agents are affected by the *status quo* they experience at the time of a decision. Agents compare an expected outcome to this *status quo* and exhibit unexpected behaviour as loss-aversion, risk-aversion and computational shortfalls.

## 4.5 Sectoral risk preferences

Specific types of economic activity can experience different preferences related to risk based on the use and benefits stream they provide to a society. For example, Ananda and Herath (2005) find that old-growth forestry is treated with much higher risk aversion than native forest removal in Australia. Through a series of surveys, the authors establish risk aversion coefficients for both activities among different groups of stakeholders. Their findings show that all groups except for tour operators are risk averse towards conserving old-growth forests while the recrationists, timber industry and tour operators exhibit risk-seeking behaviour towards other sectors (timber production and forest recreation). These differences can provide some direction in assessing the intensity of impacts on certain groups in an area. Risk-aversion towards a certain sector may be reason to prioritise projects in that sector over others less preferred.

Soane & Chmiel (2005) discuss the influence of domain and personality on risk preference. This study shows a different dynamic in personal decision making based on what part of an individual’s life is affected by a particular decision. By examining behaviour in work, personal health and personal finance, the authors show that costs and payoffs of a decision are relevant in personal, but not professional matters. Of note, perception of risk is found to have a significant role in promoting risk-averse behaviour. While not directly related to risk considerations in value transfers, this study offers background for data selection in testing whether risk preferences in two samples are similar. Where available, personal decision making data should be chosen rather than professional data.

### 4.6 Role of altruism

Altruism can play a role in valuation of public goods. Hayashi, Kotlikoff and Altonji (1991) show that complete markets without altruism do not explain behaviour sufficiently. Lusk, Nilsson and Foster (2007) find that individuals self-identifying as altruistic are more willing to pay for public good traits in goods and services (in this case environmental measures in pork-raising). This mentality shows that individuals consider more than their own preferences in purchasing and value decisions, implying a possible role for accounting for altruism in project valuation. In this paper, the authors find that self-identification of altruism is closely related to a willingness to pay for externatilities in goods and services consumed. Viewing risk reduction as an externality, this approach indicates that an altruistic culture may value risk averse policy options more than others.

## 5. Implications for cost-benefit analysis

Transferring values from previously-studied sites is an important policy tool given limited resources for direct research. In order to apply valid value transfers, site-specific information must be incorporated into a value function. A valid transfer function should account for differences in environmental services valued, population demographics and social preferences. This third category of differences can include preferences between certain goods (i.e. energy supply and environmental quality) that may vary based on an area’s history, geography and other cultural trends. These preferences can include a level of acceptable risk that varies from population to population. Accounting for differences in risk preferences between populations can help to achieve a more valid transfer of benefits values and improve cost benefit analysis.

As discussed above, several options exist for incorporating risk preferences into cost benefit analysis.

1. Risk premia in discount rates
    a. Traeger’s intertemporal risk aversion (RIRA) coefficient in the Ramsey discounting equation, which reduces discount rates with risk aversion
    b. Use of a declining discount rate over longer time horizons, or an expected net present value approach to account for increasing uncertainty
2. Gollier’s wealth premia accounts for risk preferences as well as inequality aversion in a population

The first two options (1a and 1b) are recommended by the EPA expert panel on valuation. However, the DDR/ENPV approach does not incorporate individual risks preferences and thus contributes little to improving the validity of value transfers. Therefore, we shall concentrate on Ramsey-based approaches that can allow for differentiation between sites. As the wealth premia has been shown empirically to be similar to risk aversion, RIRA appears to be the best method for capturing risk preferences from different populations. RIRA allows for differentiation between countries, which is important in value transfer and offers more methodologically than the wealth premium alone.

Including a RIRA term in a country-specific discount rate can provide a fair estimate of the social preferences at a given location and thus inform a value function for cost benefit analysis. While other options discussed above improve upon a simple unit transfer approach or non-risk-weighted value function transfer, the RIRA-specific method may yield the most accurate benefits estimate.

These tools assume a perfect world of robust data sources, which is often far from reality. Some methods for collecting data on territorial risk preferences are reviewed in Section 2 of this paper, but application of risk premia parameters may be reliant on data collection across a country or region. Decisions on how to manage heterogeneity in risk preferences to yield a single value for the parameterisation discussed above should be considered and well-documented.

## References

Ananda, J. and G. Herath. (2005). Evaluating public risk preferences in forest land-use choices using multi-attribute utility theory. *Ecological Economics , 55*, 408-419.

Andreoni, J. and C. Sprenger. (2012). Risk preferences are not time preferences. *American Economic Review , 102* (7), 3357-3376.

Arrow, K. J., M.L. Cropper, C. Gollier, B. Groom, G.M. Heal, and R.G. Newell. (2012). *How should benefits and costs be discounted in an intergeneration context?* Resources fo the Future. Washington, D.C.: Resources for the Future.

Bergland, O., K. Magnussen, and S. Navrud. (1995). Benefit transfer: testing for accuracy and reliability. *Discussion Paper #D-03/1995* (p. 21pp). Norway: Department of Economics and Social Sciences.

Bergstrom, J. C. and L.O. Taylor. (2006). Using meta-analysis for benefits transfer. *Ecological Economics , 60*, 351-360.

Brookshire, D. S. (1992). Issues regarding benefits transfer. *Association of Environmental and Resource Economists Workshop* (pp. 1-13). Snowbird, Utah: AERE.

Brouwer, R. (2000). Environmental value transfer: state of the art and future prospects. *Ecological Economics , 32*, 137-152.

Carlsson, F., D. Daruvala, and O. Johansson-Stenman. (2005). Are people inequality-averse, or just risk-averse? *Economica , 72* (287), 375-396.

Dasgupta, P. (2008). Discounting climate change. *Journal of Risk and Uncertainty , 37*, 141-169.

Freeman, M. C. (2010). Yes, we should discount the far-distant future at its lowest possible rate: a resolution of the Weitzman-Gollier puzzle. *Economics , 4* (13), online.

Fu, B.-J., C.-H. Su, Y.-P. Wei, I.R. Willett, Y.-H. Lu, and G.-H. Liu. (2010). Double counting in ecosystem services valuation: causes and countermeasures. *Ecological Resources , 26*, 1-14.

Gierlinger, J. and C. Gollier. (2008, November 3). Socially efficient discounting under ambiguity aversion. 1-26.

Gollier, C. (2002). Discounting an uncertain future. *Journal of Public Economics , 85*, 149-166.

Gollier, C. (2001). Wealth Inequality and Asset Pricing. *Review of*

*Economic Studies , 68* (1), 181-203.

Ha-Duong, M. and N. Treich. (2004). Risk aversion, intergenerational equity and climate change. *Environmental & Resource Economics, 28* (2), 195-207.

Hayashi, F., L. Kotlikoff, and J. Altonji. (1991). *Risk sharing, altruism, and the factor structure of consumption.* Federal Reserve Bank of Minneapolis, Institute for Empirical Macroeconomics. Minneapolis: Federal Reserve Bank of Minneapolis.

Howitt, R. E., S. Msangi, A. Reynaud, and K.C. Knapp. (2005). Estimating Intertemporal Preferences for Natural Resource Allocation. *American Journal of Agricultural Economics , 87* (4), 969-983.

Hsee, C.K. and E.U. Weber. (1999). Cross-national differences in risk preference and lay predictions. *Journal of Behavioral Decision Making*, 12: 165-179.

Issler, J. V. and N.S. Piqueira. (2000). Estimating relative risk aversion, the discount rate, and the intertemporal elasticity of substitution in consumption for Brazil using three types of utility function. *Brazilian Review of Econometrics , 20* (2), 201-239.

Kahneman, D. and A. Tversky. (1979). Prospect theory: an analysis of decision under risk. *Econometrica , 47* (2), 263-292.

Liu, E.M. (2013). Time to change what to sow: risk preferences and technology adoption decisions of cotton farmers in China. *The Review of Economics and Statistics*, 95: 4: 1386-1403.

Lusk, J. L., T. Nilsson, and K. Foster. (2007). Public preferences and private choices: effect of altruism and free riding on demand for environmentally certified pork. *Environmental & Resource Economics , 36*, 499-521.

Mazzocco, M. and S. Shivani. (2007). Testing efficient risk sharing with heterogenous risk preferences. *California Center for Population Research On-Line Working Paper Series , 15* (7).

Mirrlees, J. A. (1971). An exploration in the theory of optimal income taxation. *Review of Economic Studies , 38* (1), 175-208.

Navrud, S. and R. Ready. (2006). Review of methods for value transfer. In S. Navrud, & R. Ready, *Environmental value transfer: issues and methods* (pp. 1-25). The Netherlands: Springer.

OECD. (2006). Benefits transfer. In OECD, *Cost-benefit analysis and the environment: recent* *developments.* Paris: OECD Publishing.

Ready, R., S. Navrud, B. Day, R. Dubourg, F. Machado and S. Mourato. (2004). Benefit transfer in Europe: how reliable are transfers between countries. *Environmental and Resource Economics* .

Rosenberger, R. S. and J.B. Loomis. (2003). Benefit Transfer. In P. A. Champ, K. J. Boyle, & T. C. Brown, *A Primer on Nonmarket Valuation* (Vol. 3, pp. 445-482). Netherlands: Springer.

Rosenberger, R. S. and T.D. Stanley,. (2006). Measurement, generalisation, and publication: sources of error in benefit transfers and their management. *Ecological Economics , 60*, 372-378.

Silva, P. and S. Pagiola. (2003). *A review of the valuation of environmental costs and benefits in World Bank projects.* The World Bank, Environmental Department. Washington, D.C.: The World Bank.

Snydor, J. (2010). (Over)insuring modest risks. *American Economic Journal: Applied Economics*, 2: 177-199.

Soane, E. and N. Chmiel. (2005). Are risk preferences consistent? The influence of decision domain and personality. *Personality and Individual Differences , 38*, 1781-1791.

Spash, C. L. and A. Vatn. (2006). Transferring environmental value estimates: issues and alternatives. *Ecological Economics , 60*, 379-388.

Stern, N. H. (1976). On the specification of models of optimum income taxation. *Journal of Economic Theory , 6* (1-2), 123-162.

Stern, N. H., et. al. (2007). *The Stern Review of th economics of climate change.* Cambridge: Cambridge University Press.

The Economist Online. (2011, November 1). Focus: Brazil. *The Economist* , p. Online.

Traeger, C. P. (2009). Recent Developments in the Intertemporal Modeling of Uncertainty. *The Annual Review of Resource Economics , 1*, 261-85.

Weil, P. (1992). Equilibrium asset prices with undiversifiable labor income risk. *Journal of Economic Dynamics and Control , 16*, 769-789.

Weitzman, M. L. (2011). Fat-tailed uncertainty in the economics of catastrophic climate change. *Review of Environmental Economics and Policy , 5* (2), 275-292.

Weitzman, M. L. (2007). The Stern Review of the economics of climate change. *Journal of Economic Literature , 45* (3), 703-724.

Weitzman, M. L. (1998). Why the far-distant future should be discounted at its lowest possible rate. *Journal of Environmental Economics and Management , 36* (3), 201-208.

Weitzman, M. (2009). On Modeling and Interpreting the Economic Effects of Catastrophic Climate Change. *Review of Economics and Statistics , 51* (1), 1-19.

Wilson, M. A. and J.P. Hoehn. (2006). Valuing environmental goods and services using benefits transfer: the state-of-the art and science. *Ecological Economics , 60*, 335-342.