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INCORPORATING ECOSYSTEM SERVICES INTO NATIONAL ACCOUNTING: THEORY AND PRACTICE

By

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A Dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Economics

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To Eneko, Mertxe,

Mikel and Luisa

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SUMMARY

This dissertation deals with the incorporation of non-market ecosystem services into the national accounts from a theoretical and empirical point of view. The overall purpose is to show that it is possible to estimate the economic value of these services in a consistent way with market goods and services. This requires the use of exchange values (prices times quantities).

Terrestrial ecosystems, and more specifically forests, provide goods and services that are relevant to society because they generate human well-being. Significant efforts have been made during the last decades in order to extend national accounts to integrate ecosystem services. The System of Environmental and Economic Accounting – Experimental Ecosystem Accounting has been recently presented. Some research projects have also been launched in order to apply ecosystem accounting in several countries. However, some of these research efforts are not linked with theoretical literature on green national accounting.

Valuation of non-market ecosystem services has been done traditionally using non-market valuation techniques, which mainly focus on the demand side. The demand function is interpreted as the probability that individuals would be willing to pay a given amount of money for the consumption of an ecosystem service. This allows Hicksian variations to be estimated. These measures are mainly used in cost-benefit analyses, but conventional systems of national accounts reject the use of any welfare measures, proposing the use of exchange values instead. Thus, if we seek to integrate non-market ecosystem services into the national and ecosystem accounting, we need to distinguish the part that could be internalized in terms of exchange values. Indeed, theoretical studies also point out the need to focus only on this part.

The theoretical model that we present in the third chapter of this dissertation confirms that non-market values can be incorporated into national accounting using (*simulated*) exchange values, and not consumer surplus or any other Hicksian variation. We propose the Simulated Exchange Value (SEV) method and detail the implications of these results for applied accounting. It is precisely from this empirical perspective that, in the fourth chapter, we compare aggregated recreation values for stone pine and cork oak forests in Spain obtained by using compensating variation (a type of Hicksian

variation) and by using simulated exchange value (maximum benefits from a potential market) measures. The latter measures account for 35 to 51% of the former values. This is because when a price is set, only part of the population would pay that price. We use a choice experiment in order to estimate the demand function of public recreation in the above-mentioned forests, and we test for construct validity of willingness to pay values as the estimation of compensating variation and the simulated exchange value relies on the validity of the experiment. Results show convergent validity between a choice and a ranking recoded as a choice format in an experiment with three alternatives plus status quo. We also find significant differences between two payment vehicles (increased trip expenditures and entrance fee) that are included simultaneously in the choice sets.

Biodiversity conservation is another example of non-market ecosystem service. In the fifth chapter we also determine aggregated values for the conservation of the Iberian Lynx in Spain. Again, we compare the compensating variation measure associated with a program for the conservation of this endangered species with the results obtained from the Simulated Exchange Value method for the same program. Unlike in the previous chapter, here we use a contingent valuation analysis to estimate the demand function and analyze whether the existence of different and possibly conflicting preference orderings affects the valuation function of the respondents' WTP and the estimated aggregated values. Results show that simulated exchange values account for 55 to 64% of the compensating variation values.

From another angle, another contribution of the dissertation is the valuation of net accumulation of forest capital. The theoretical literature on green accounting identifies the Net National Product measured by the national accounts as by far the most relevant indicator of social welfare and points out that this indicator can be measured using information for the current year as forests are made up of a single species whose growth is instantaneous. We show that, in the case of the forestry sector, the results previously obtained only hold true if there is a single species whose growth can be approximated by an exponential function with constant decay. If there are different types of species, the information of the year does not suffice, as there are future values. The demonstration of this result requires the application of Volterra integral equations, which have been traditionally utilized in the well-known 'vintage' models.

RESUMEN

La presente tesis doctoral trata sobre la incorporación de los servicios del ecosistema de no mercado a la contabilidad nacional desde un punto teórico y aplicado. El objetivo principal de esta investigación es mostrar que es posible estimar el valor económico de estos servicios consistentemente con la forma en que se valoran los bienes y servicios de mercado. Esto requiere utilizar valores de cambio (precios por cantidades).

Los ecosistemas terrestres, y en particular los bosques, ofrecen bienes y servicios que son relevantes para la sociedad en la medida en que generan bienestar. En las últimas cuatro décadas se han venido realizando importantes esfuerzos con el objetivo de ampliar las cuentas nacionales para integrar a los servicios del ecosistema. El Sistema de Contabilidad Ambiental y Económica –Contabilidad Experimental de los Ecosistemas ha sido presentado recientemente. Igualmente, se han puesto en marcha algunos proyectos de investigación con el objetivo de aplicar la contabilidad de los ecosistemas en determinados países. Sin embargo, algunos de estos esfuerzos de investigación no están vinculados a la literatura teórica en torno a la contabilidad verde.

La valoración de servicios del ecosistema de no mercado se ha realizado tradicionalmente mediante técnicas de valoración de no mercado, es decir, centrándose en el lado de la demanda. La función de demanda se interpreta como la probabilidad de que los individuos estén dispuestos a pagar una determinada cantidad de dinero por el consumo de un servicio del ecosistema. Esto permite calcular diferentes variaciones Hicksianas. Estas medidas son principalmente utilizadas para la realización de análisis de costo-beneficio. Sin embargo, los sistemas de cuentas nacionales convencionales rechazan el uso de cualquier medida de bienestar, proponiendo el uso de valores de cambio. Por lo tanto, si el objetivo es integrar los servicios de los ecosistemas de no mercado en la contabilidad nacional, necesitamos distinguir la parte que podríamos internalizar en términos de valor de cambio. De hecho, los estudios teóricos señalan la necesidad de centrarse únicamente en esta parte.

El modelo teórico que presentamos en el tercer capítulo de esta tesis confirma que los valores de los ecosistemas de no mercado pueden ser integrados en la contabilidad nacional utilizando valores de cambio *(simulados)* y no el excedente del consumidor u alguna otra variación Hicksiana. Proponemos el método del Valor de Cambio Simulado

(VCS) y detallamos las implicaciones de estos resultados en lo que se refiere a la contabilidad aplicada. Es precisamente desde esta perspectiva empírica desde la que, en el cuarto capítulo de la tesis, comparamos los valores agregados para el uso recreativo en bosques de pino piñonero y alcornoque en España. Para ello, utilizamos dos medidas: la variación compensatoria (un tipo de variación Hicksiana) y el valor de cambio simulado (es decir, los máximos beneficios que se pueden obtener en un mercado potencial). Las medidas obtenidas utilizando este último método representan entre un 35 y un 51% de los valores obtenidos utilizando la variación compensatoria. Esto se debe a que cuando se establece un precio, únicamente una parte de la población estaría dispuesta a pagar dicho precio. En este ejercicio, utilizamos un experimento de elección para estimar la función de demanda del uso recreativo de los bosques mencionados. No obstante, la estimación de ambas medidas depende de la validez del experimento, por lo que también realizamos dos análisis de validez convergente de los valores de la disposición a pagar. Los resultados muestran convergencia entre un formato de elección y otro de ordenación recodificado como una elección en un experimento con tres alternativas más el status quo. Por otro lado, se identifican diferencias entre dos vehículos de pago (pago de una entrada e incremento en los gastos de viaje) incluidos simultáneamente en el mismo escenario de elección.

La conservación de la biodiversidad representa otro ejemplo de servicio del ecosistema de no mercado. En el quinto capítulo de la tesis se determinan los valores agregados para la conservación del lince Ibérico en España. De nuevo, comparamos la variación compensatoria asociada a un programa de conservación de esta especie amenazada con los resultados obtenidos al aplicar el método del Valor de Cambio Simulado para el mismo programa. A diferencia del capítulo anterior, en este caso realizamos un estudio de valoración contingente para estimar la función de demanda. También analizamos si la existencia de motivaciones diferentes, y posiblemente en conflicto, a la hora de pagar afecta a la función de valoración de la disposición a pagar de los encuestados y a los valores agregados estimados. Los resultados muestran que los valores de cambio simulados representan entre un 55 y un 64% de los valores obtenidos utilizando la variación compensatoria.

Desde otra perspectiva, otra aportación de la tesis es la valoración de la acumulación neta del capital forestal. La literatura teórica sobre la contabilidad nacional verde identifica el Producto Nacional Neto medido por las cuentas nacionales como el indicador más relevante del bienestar social y señala que dicho indicador puede ser estimado utilizando únicamente valores del año en curso, ya que los bosques están formados por una única especie cuyo crecimiento es instantáneo. En esta tesis, mostramos que, en el caso del sector forestal, los resultados previamente obtenidos siguen siendo válidos si existe una única especie cuyo crecimiento puede ser aproximado por una función exponencial con un decaimiento constante. Si existen diferentes tipos de especies, la información del año no es suficiente, ya que hay valores a futuro. La demostración de este resultado requiere la aplicación de ecuaciones de Volterra, las cuáles han sido tradicionalmente utilizadas en los conocidos como modelos 'vintage'.

ACRONYMS

AAS	Agroforestry Accounting System
ANI	Domestic animals
ASC-REC	Constant specific to forest visit alternatives
BEA	Bureau of Economic Analysis
CBA	Cost-Benefit Analysis
CH	Choice
CICES	Common International Classification of Ecosystem Services
CREEA	Compiling and Refining Environmental and Economic Accounts
CV	Compensating variation
DHSS	Dasgupta-Heal-Solow-Stiglitz model
EEA/EAF	Methodology on the Economic Accounts for Agriculture and Forestry
ENT	Entrance fee
ENTP	Entrance fee per person
ESA	European System of Accounts
EUROSTAT	Statistical Office of the European Communities
EV	Equivalent Variation
EXP	Increased trip expenditures
EXPP	Increased trip expenditures per person
FR	Homo Strategicus
FUND	Contribution to a fund
FUND1	Contribution to a fund managed by Non-Governamental Organizations
FUND2	Contribution to a fund managed by Non-Governamental Organizations with a guarantee that funds would be reimbursed if not enough money were collected
GDP	Gross Domestic Product
GNP	Gross National Product

GNNP	Green Net National Product
HE	Homo Economicus
HEP	Model differentiating between Homo Economicus and Homo Politicus while eliminating the free-riding effect
HEPFR	Model differentiating between Homo Economicus and Homo Politicus
HP	Homo Politicus
HP_{sh}	Homo Politicus with shared responsibility
HP_{so}	Homo Politicus with sole responsibility
HSSFR	Model differentiating between Homo Economicus, Homo Politicus with sole responsibility, Homo Politicus with shared responsibility and Homo Strategicus
IEEAF	Integrated Environmental and Economic Accounting for Forest
IEESA	US Integrated Environmental and Economic Satellite Accounts
IIA	Independence of irrelevant alternatives
INS	Infrastructures for recreation
IMF	International Monetary Fund
IPCC	Intergovernmental Panel on Climate Change
ISWGNA	Intersecretariat Working Group of National Income Accounts
IUCN	International Union for Conservation of Nature
MA	Millennium Ecosystem Assessment
MUSH	Mushroom collecting
NDP	Net Domestic Product
NGO	Non-Governamental Organizations
NIPA	National Income and Product Accounts
NNP	Net National Product
NOAA	National Oceanic and Atmospheric Administration
NPV	Net Present Value
NVA	Net Value Added
OECD	Organization for Economic Cooperation and Development

RC	Recoded choice
RECAMAN	Renta y Capital de los Montes de Andalucía
REDD+	Reducing Emissions from Deforestation and Forest Degradation
RP	Revealed preferences
SEEA	System of Environmental-Economic Accounting
SEEA-CF	SEEA-Central Framework
SEEA-EEA	SEEA-Experimental Ecosystem Accounting
SEEA-ENERGY	System of Environmental-Economic Accounting for Energy
SEEA-WATER	System of Environmental-Economic Accounting for Water
SEV	Simulated Exchange Value
SNA	U.N. System of National Accounts
SP	Stated preferences
TAX	Increased income tax
TAX1	Additional increased income tax
TAX2	Referendum for an increased income tax
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TREE	Type of forest
UK NEA	UK National Ecosystem Assessment
UNCED	United Nations Conference for Environment and Development
UNEP	United Nations Environment Programme
UNSD	United Nations Statistical Division
UNSO	United Nations Statistical Office
VA	Value Added
VCS	Valor de Cambio Simulado
WAVES	Wealth Accounting and the Valuation of Ecosystem Services
WTA	Willingness to Accept
WTP	Willingness to Pay

NOTATION

A	Aggregate consumption of renewable resources (fish, forest biomass)
C	Consumption of market goods and services
D	Land clearing (deforestation)
E	Expectation
F	Forest management and forestland
G	Growth of all vintages
I	Agricultural land
K	Stock of man-made capital
L	Land
M	The accumulated growth in wood of each tree at the time of felling
N	Households' collection of non-timber products
P	Stock of pollution
Q	Current number of visits (chapter 4) or current population (chapter 5)
R	Stock of renewable resources (fish, forest biomass) and revenue function
S	Consumer surplus
T	Time of felling trees
U	Social utility
V	Systematic component
W	Logging
X	Vector of observable variable
Y	Production function
Z_W	Production of logs (wood)
Z_N	Production of non-timber products
Z_D	Newly cleared land
b	Species
c	Costs
e	Emissions of air pollutants
g	Natural growth in the stock of renewable resources (fish, forest biomass)

h	Harvest
i	Net investment flow
j	Alternative (visit a forest or preservation of the Iberian Lynx)
k	Set of attributes
l	Individual facing the choice experiment or contingent valuation survey
m	Growth function of one tree planted at time *s* in any moment in time *t*
p	Monetary price of consumption
p_i	Cost of buying an extra unit of capital good or the market price of investment
p_{Ω}	Market price of maintenance (defensive) expenditures
p_N	Market price of non-timber products
p_w	Market price of the forest biomass (wood) as a factor of production
p_h	Market price (cost) of extracting wood
p_{F_x}	Marginal revenue product of the amount of growth produced at time *t*.
p_x	Market price (cost) of planting trees
p_b	marginal revenue product of the species planted at time *t*
p_A	(*Simulated*) price of non-market ecosystem services
p_j	Price for accessing the forest and price set for the conservation program
q	Quantity of visits to a forest and quantity of contributors to the program
r	Discount rate or rate of return
u	Choice situation
x	Amount of growth produced at time *s*
y	Other alternative against visiting a forest or preserving the Iberian Lynx
w	Capital income obtained by the forest owner
$aY(K)$	Degradation due to the economic activity
Bid_l^u	Upper Bid
Bid_l^d	Lower Bid
pv	Payment vehicle

$\Pr$	Probability
$\ln L(\theta)$	Log-likelihood
PP	Production in progress
α	Rate of discount of the utility flow
β	Parameter
γ	Degradation from natural causes
δ	Depreciation rate
ε	Random errors
ξ	Fishing cost
θ	Parameters
λ	Shadow values
μ	Scale factor
ρ	Costs of maintenance
ς	Cost of planting
σ	Decay rate of carbon
ℓ	Current labor force
υ	Number of entrance fees purchased
$\phi(i)$	Cost of investment
$\psi(h)$	Cost of extracting forest biomass
$\omega(\upsilon)$	Cost of the payment of the entrance fee
Ψ	Number of draws from θ
Π	Profits
Φ	Production possibilities set
Ω	Level of improvements or maintenance
q_{hs}	Quantity of timber felled
$c(q_h(T))$	Extraction costs
π_{jd}	Conditional probability
p_p	Price for not felled standing timber
s_j	Average age of class *j*

s_d	Age of the present diameter class *d*
N_d	Number of diameter-ages *d*
PP_i	Production in progress at the beginning of the accounting period
PP_f	Production in progress at the end of the accounting period
q_{pjt}	Not felled timber for each diameter-class
PP_u	Timber extracted from the forest
PP_e	Total stock entrance or equivalently natural gross commercial growth
PP_{rem}	Revaluation of timber during the period
q_{hj}	Annual commercial gross growth of the diameter-class *j* that has not been extracted

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CHAPTER 1. INTRODUCTION

1. INTRODUCTION

A large body of research has been concerned with the issue of the appropriate use and design of the national accounts over the last decades (Repetto et al., 1989; Hartwick, 1990; Mäler, 1991; Aronsson and Löfgren, 1998a,b; Nordhaus and Kokkelenberg, 1999; Perrings and Vincent, 2003; Aronsson et al., 2004; Lange, 2007; Dasgupta, 2009; Atkinson, 2010). The growing interest among economists in environmental matters has manifested itself in the application of economics to the environment. The field of environmental economics has gone a long way in that regard, but it still has important contributions to make. Mainstreaming the value of natural capital into policy decision-making is vital, as the consumption and enjoyment of goods and services that nature provides contribute directly and indirectly to human well-being.

There is recognition that natural resources and associated ecosystem services should be incorporated into the national accounts from an economic standpoint. Natural capital accounting therefore will help giving a more complete picture of economic activity and the interactions between agents. However, the fact of setting a price on nature has often been criticized. Just because some ecosystem services are not traded in the marketplace does not mean that their monetary value cannot be estimated. Their economic valuation has indeed been subject of debate in recent decades, and several non-market valuation techniques have been developed.

From a theoretical point of view, there is also a large body of literature on how national accounting should be modified to accurately measure the income generated by a country in a year. One of the most relevant results is that, under certain assumptions, Net National Product (NNP) is the appropriate indicator of economic progress and social welfare, as it summarizes the necessary information to move the economy towards the optimal path. The need of expanding this concept to take into account natural resources has resulted in what is known as green NNP (GNNP).

Accordingly, whatever perspective – accounting, valuation or theoretical – we take, the assessment of the natural environment leads to a better understanding of its role in the economy. Still, there are fundamental differences between them when it comes to the valuation principles and criteria applied, especially for the case of non-market ecosystem services.

If the goal is to include these services in the national accounts in a consistent way with market goods and services, exchange values should be calculated, as recommended by the System of National Accounts (SNA). However, most applied non-market valuation studies fall under the scope of the cost-benefit analysis (CBA) and calculate consumer surplus or any other Hicksian variation measure; while theoretical frameworks and conventional systems of national accounts point out that these measures must not be used[1].

The general purpose of this dissertation is to demonstrate the possibility of consistently integrating market and non-market ecosystem services into the national accounts, regardless of whether we are dealing with purely theoretical models or with practical applications. With a few exceptions, the task of knitting both perspectives together lay ahead. This research aims to offer an approach in this regard.

One specific objective consists in expanding conventional optimal control models in order to incorporate non-market *amenities* in the same way as market goods and services and to adequately measure the stock of natural capital. From an applied perspective, the main focus is on showing that it is also possible to estimate exchange values for two of the most common non-market ecosystem services, which are public forest recreation and biodiversity conservation.

Overall, biodiversity valuation may be tacklet by using either dynamic bioeconomic models (or ecological economic modeling) or preference-based methods. The former are charactetized by combining biophysical and economic components, namely population dynamics and economic mathematical models[2]. Most applications have been

[1] Thus, although using CBA would be, ideally, the most appropriate, we cannot use it if we wish to be consistent with the proposal of national accounting systems, as we will see below.

[2] Economic problem is constrained by the population dynamics of exploited species, which are described by the equations of motion of the system (Perrings, 2001). This involves integrating Lotka-Volterra equations, as dynamics of biological systems are nonlinear (see also Pascual and Muradian (2010)).

designed in the context of fisheries management and agricultural policies (Larking et al., 2011; Mouysset et al., 2011, 2015; Pereau et al., 2012; Doyen et al., 2013; Cissé et at., 2013). On the contrary, preference-based valuation methods rely on the assumption that values arise from individuals' preferences and their expression through actual or hypothetical markets. In this dissertation, we will use the latter approach in order to estimate the economic value of a program to conserve the Iberian lynx in Spain.

1.1. Background

At the end of the 17th century, W. Petty developed what might be considered the early version of an accounting system at national level. He supported the use of statistical techniques for measuring social phenomena and began analyzing the method of political arithmetic. As early as in the 18th century, physiocrats, and fundamentally F. Quesnay, advanced in the understanding of the interdependence between economic sectors, as reflected in the Tableau Économique. Nevertheless, the increase in foreign trade since the Industrial Revolution in the second half of the century laid the foundations for the future transformation of the national accounts.

This progress notwithstanding, the major advances went hand in hand with Keynesian economics after the World War II. J.M. Keynes developed detailed estimates of national income and expenditure in Great Britain, giving rise to the 'modern' national accounts. R. Stone later worked on double-entry bookkeeping methods and conducted several studies on the measurement of income and consumer demand in the UK for which he received the Nobel Prize in 1984. He also focused on socio-demographic accounts, developing the social accounting matrix.

Based on the double-entry bookkeeping method, national accounting systems emerged around the efforts to measure aggregated economic activity. Although the first formal national accounts were published in the United States, soon there were also important advances in Europe. That is why the United Nations began preparing the groundwork for the future development of the SNA.

National accounts consist of two sets of accounts: current accounts and assets accounts (capital balance). The former provide information on monetary transactions related to the production and use of goods and services and on the distribution and redistribution of income from productive activities (ISWGNA, 2009: 3). Gross

Domestic Product (GDP) is the main aggregate measure in these accounts. Asset accounts describe the changes in the stock of an asset, as well as the monetary information on stocks of productive assets (ISWGNA, 2009: 331). Both accounts contain two common concepts: consumption of fixed capital (depreciation) and gross capital formation (gross investment). Thus, subtracting them from GDP yield conventional Net Domestic Product (NDP).

Even so, as regards ecosystems, the SNA only measures the commercial net value added (NVA) generated by a forest, i.e. the value of commercial extraction, net of intermediate and man-made capital consumption. Differences experienced during the period by non-redeemable man-made capital and natural capital are thus left out (Caparrós et al., 2003). This measurement is at odds with the definition of income issued by J. Hicks: "the maximum value which a person can consume during a week, and still expect to be as well off at the end of the week as he was in the beginning" (Hicks, 1946: 172). That is, the maximum amount a person can consume while keeping overall capital intact because "as long as net investment is kept non-negative, the capital remains intact" (Aronsson and Löfgren, 1998a: 211). A precedent to this concept can be found in I. Fisher: "the income from any instrument is thus the flow of services rendered by that instrument. The income of a community is the total flow of services from all of its instruments" (Fisher, 1906: 101)[3]. The correct measurement of total income therefore involves measuring Hicksian income. Advances have been made toward achievement of this goal over the last two decades, but there is still some way to go. The Agroforestry Accounting System (AAS) (Campos, 1999; Caparrós et al., 2003) is, as discussed below, an example in this sphere.

Although GDP is the best-known indicator of aggregated economic activity (this concept coincides with Gross National Product (GNP) in a closed economy), the theoretical literature recommends the use of NNP, a concept closely related to Hicksian income. The goal of theoretical models is to determine the changes that should be made to the national accounts such that they can appropriately measure NNP, as well as its contribution to social well-being. This has resulted in what has come to be called ‘social accounting’ (Turner et al., 1994; Aronsson et al., 1997).

[3] Lindahl advanced Fisher's concept understanding income as the sum of consumption and net increment of wealth over a period (Weitzman, 2003).

The welfare interpretation of NNP comes originally from the seminal paper by Weitzman (1976). He also derived the result that the Hamiltonian for a general equilibrium model could be interpreted, under certain assumptions, as equivalent to the NNP measured by the national accounts. The corollary is that the economy needs to maximize NNP in order to follow the optimal path[4]. Subsequent developments of the Weitzman's approach have extended NNP to cover natural resources (Solow, 1986; Hartwick, 1990, 2001; Mäler, 1991) and the negative effects of pollution (Brock, 1977; Hartwick, 1990; Tahvonen and Kuuluvainen, 1993). Terrestrial ecosystems, ans more specifically forests, have also been addressed, albeit to a minor extent (Vincent, 1999a; Cairns, 2001, 2003; Matero and Saastamoinen, 2007; Caparrós, 2010).

Apart from these theoretical studies, there are also many applications oriented towards estimating the economic value of ecosystems at national or international level (Repetto et al. (1989) pioneered this approach for Indonesia's forests), and different research projects put into practice. We do not wish at this point to go so much into this issue, as we will explain it throughout the second chapter.

Nevertheless, this brings to the forefront, in what ways and to what extent, the value of natural resources can be estimated in monetary terms. Monetary valuation of environmental goods has become the subject of a large body of research over the last few decades. In this sense, economists interested in environmental management have been developing a range of techniques to assign monetary values to the environment. Considering that economic valuation techniques are intended to assess whether the costs of certain project or policy are justified in terms of the benefits generated, the main purpose of environmental valuation consists in including environmental concern in the CBA[5] from a monetary point of view.

[4] There are, however, some authors (Dasgupta and Mäler, 2000; Arrow et al., 2003a,b; Dasgupta, 2009) criticizing the use of NNP to evaluate social welfare. As the economy does not follow an optimal path, it is better to implement an economic program so as to make sure that the economy keeps evolving.

[5] CBA arose during the late 1930s from an intense debate between economists L. Robbins, Hicks and N. Kaldor. While it seemed a well-accepted principle that the capacity of enjoyment was equal for all individuals, Robins argued that it was not possible to make comparisons between their utilities. Kaldor claimed that, even if this were true, it was not the relevant point. The key was to know whether the implementation of a policy increased total real income so that it was possible to give a Pareto optimal allocation (Zerbe, 2013). Hicks agreed this approach, giving rise to the Kaldor-Hicks criterion.

In this context, two concepts emerge: willingness to pay (WTP) and willingness to accept (WTA). The former represents the maximum amount of money that an individual is willing to pay for a marginal change in the provision of a good or service (Atkinson, 2010). The latter is the minimum amount of compensation that an individual is willing to accept to forego a marginal change. At all events, both measures are monetary measures of welfare changes and reflect individuals' preferences (Kriström and Johansson, 2015). Considering that preferences are represented by utility functions, the economic value of a good or service may alternatively be defined as the change in the utility derived from a marginal change.

Hicks (1941, 1943) proposed two WTP measures in order to estimate welfare changes in monetary terms: Equivalent Variation (EV) and Compensating Variation (CV). The former determines the change in consumers' income that would lead them to the same utility level as that generated by a change in market prices. The latter estimates how much consumers' income need to increase (or decrease) in order to get them back to the same utility that they had before a change in market prices (note that, compensation takes place after the price change, so CV uses the existing prices after the change). Another tool for measuring welfare changes is consumer surplus, i.e. the difference between consumers' WTP and the price they actually pay. It is closed to EV and CV if the wealth effect is small (in this case we would refer to the Marshallian consumer surplus), and an exact measure of EV and CV if the utility is also quasilinear.

An important point to bear in mind is that individuals not just derive utility for the mere fact of using the good or service at the current moment, but for knowing that they will have the possibility of using it in the future. This means that total economic value (TEV)[6] consists of use values and passive use values. Use values are divided into direct use values, indirect use values and option values; while passive use values are divided into existence values, altruistic values and bequest values (Pearce and Turner, 1990). Non-market valuation techniques, a matter to which we will devote an important part of this dissertation, appear as a tool for estimating these values. Roughly speaking, these techniques consist of revealed preference methods and stated preference methods. The former only capture use values, while the latter capture use and passive use values.

[6] In environmental economics, TEV is refers to the aggregated value of the benefits provided by ecosystem services in a given state (Pascual and Muradian, 2010).

1.2. Motivation

The main motivation behind this dissertation is the need to bridge the gap between theoretical analyses, applied studies and standards proposals of accounting systems, especially when it comes to forests. This requires setting down clear guidelines.

A first important difference concerns valuation rules. While one should be able to measure the economic value of nature as if it were other form of capital, some ecosystem goods and services are not traded in the marketplace, thereby not making it possible to directly observe market prices. In those cases, central normative national accounts frameworks themselves aim at using market price equivalents as the first option. But for certain ecosystem *amenities*, such as public forest recreation or biodiversity conservation, there are no similar markets. In these circumstances, the first temptation is to use consumer surplus (or more precisely, Hicksian variations) obtained with non-market valuation techniques. Nevertheless, national accounting systems oppose the use of consumer surplus and theoretical studies on forests also take it out when relating the linearized current-value Hamiltonian to GNNP. While Hicksian variation estimates are relevant in CBA, for national accounts one needs to distinguish the part that could be internalized in terms of exchange values. Cairns (2003), in fact, proposes focusing on this part, as this is the only one consistent with current estimates in the conventional SNA.

An interesting proposal that is linked up with the above-mentioned idea is the application of the Simulated Exchange Value (SEV) method. It was initially proposed in Caparrós (2000) and later empirically applied in Caparrós et al. (2003), Campos and Caparrós (2006) and Caparrós et al. (2015). This methodology is intended to obtain the (*simulated*) economic value of ecosystem services for which there are neither observable market prices nor similar markets. In doing so, it simulates the whole market (demand, supply and competitive environment) in order to estimate the (*simulated*) market price that the forest owner would set for the service if it were internalized.

We do believe that, for the time being at least, it is the best method to estimate the economic value of non-market ecosystem services. Being coherent with the guidelines laid down by accounting systems, it also makes it possible to integrate market and non-market ecosystem services uniformly from the theoretical point of view. Interestingly enough, this method also guarantees this consistency when it is empirically applied.

Another issue to which we need to turn out our attention in this dissertation is the treatment of future values when estimating the stock of natural capital, which is by no means negligible. Most theoretical literature on forest accounting assumes that forests are made up of a single species whose growth is instantaneous (Cairns 2001, 2003). That is, the age-class of the forest is ignored, thus obtaining the result that it is enough to use information of the current year for calculating the current-value Hamiltonian and GNNP (Vincent, 1999a). However, this result only holds for strong and probably unrealistic assumptions.

Vincent (1999b) and Caparrós et al. (2003)[7] discuss the role of the age-class structure of the forests, although in partial equilibrium framework (still assuming that there is one type of species). Under these conditions, the use of values of the current year remains possible. Only if different species existed would theoretical studies integrate forward-looking values (see, e.g. Caparrós (2010)). In the third chapter of this dissertation, we estimate the contribution of forest biomass in terms of future values. This implies having a problem with an integral state equation of the Volterra type. The main implication of this special type of state equations is that they permit aggregating the growth of the different vintages in the forest. However, they make the problem more complex, as the equation cannot be simplified to the standard state equation by taking the time derivative (Vinokurov, 1969; Kamien and Muller, 1976). Although Volterra equations have been generally used to analyze durable goods in 'vintage' models (Schmalensee, 1979; Muller and Peles, 1990), in this dissertation we stress that this approach can be extended for modeling the growth of forests within a green accounting framework. Hence, this proposal opens up the possibility of more realistic scenarios by applying tools that have not been applied before in this field.

1.3. Preview of results

We express the need to agree upon a new methodology reflecting the contribution of ecosystem services with and without market prices, as well as without similar markets. This requires purging consumer surplus and using exchange values instead. We consider the SEV method to be the best choice to do so.

[7] In fact, Caparrós et al. (2003) calculate the stock values following this recommendation.

The second chapter of this dissertation makes an in-depth review of previous literature on national accounting from a theoretical, empirical and accounting point of view and focuses on terrestrial ecosystems. If follows from this analysis that there are differences between these perspectives, for example on how the value of non-market ecosystem services must be estimated. Therefore, it is relevant to discuss these divergent positions and to make proposals for resolution.

In the third chapter, we focus on the theoretical part by presenting a general equilibrium model that makes it possible to expand Weitzman's results to incorporate non-market ecosystem services. In this way we show that it is possible to integrate these services into the GNNP definition and, most importantly, to include them in a consistent way with market goods and services. In order to be coherent with the proposal of national accounting systems, exchange values must be used. To that end, we simulate the 'quasi-market' prices for forest *amenities* such as public recreational use. We thus provide the theoretical basis for the SEV method. As explained below, this method requires knowledge of the competitive environment of non-market ecosystem services. Hence we characterize exchange values for perfect competition, monopoly and monopolistic competition.

Apart from this, we also demonstrate that the possibility of defining GNNP by using only values of the current year is only available under certain restrictive conditions. It would apply, for instance, if trees grow instantaneously and belong to the same species, as assumed by previous studies on forests (Vincent, 1999a; Cairns, 2001, 2003). We show that this result also remains certain when the single species grows following a constant decay exponential function and trees are not felled. However, these two scenarios are unrealistic. In order to approach this problem in a more realistic manner, we prove that when reforestations are carried out with several species growing differently, there are forward-looking values within the GNNP calculation. Thus, we develop an approach that advances better understanding on the role of future values in the estimation of net accumulation of forest capital.

In this dissertation we also use the SEV method to estimate the aggregated values of two non-market ecosystem services, which are public recreation (the fourth chapter) and biodiversity conservation (the fifth chapter). In latter, we focus on a conservation program for the Iberian Lynx in Spain, as it is one of the most endangered feline species originally native to the Iberian Peninsula.

In order to simulate exchange values, the demand function must be estimated using non-market valuation techniques. In the fourth chapter, we present a choice experiment and we analyze its validity by testing the convergent validity of two elicitation formats (ranking and choice recoded as a choice) and two payment vehicles (entrance fee and increased trip expenditures). Results show convergent validity between elicitation formats in an experiment with three alternatives plus status quo; while divergent results when including both payment vehicles simultaneously in the choice sets. We also estimate aggregated recreation values. Choice experiments application usually focus on compensating variation (a welfare measure in line with consumer surplus), but we extend this analysis by comparing this measure with the simulated exchange value measure. Results indicate that the latter measure accounts for 35 to 51% of the former values.

In the fifth chapter, we present a contingent valuation study for the conservation of the Iberian Lynx and we analyze it in terms of preference ordering. Individuals may be motivated by reasons other than the direct use of the good or service in scrutiny. The only study formalizing the differentiation between consumers (or *Homo Economicus*) and citizens (or *Homo Politicus*) in the field of environmental valuation can be found in Nyborg (2000). She goes beyond and differentiates *Homo Politicus* between *Homo Politicus with sole responsibility* and *Homo Politicus with shared responsibility*. In addition to the two above-mentioned types of individuals, we introduce another one, called *Homo Strategicus*. Results show that there are some differences between them when it comes to facing with a contributing scenario. After identifying empirically this triple consumer-citizen-strategic distinction, we also estimate aggregated values of the biodiversity conservation program. As in the fourth chapter, we compare compensating variation and simulated exchange value measures. Results indicate that the latter measures account for 55 to 64% of the former values, depending on the type of respondent considered.

CHAPTER 2. A REVIEW OF THE LITERATURE ON NATIONAL AND ECOSYSTEM ACCOUNTING

2. A REVIEW OF THE LITERATURE ON NATIONAL AND ECOSYSTEM ACCOUNTING

In this second chapter we conduct a comprehensive literature review in order to find out common points and contrasting elements between accounting-related theoretical studies and national accounting systems. We seek, however, as the title indicates, to make further progress in answering how natural resources can be integrated within these approaches. This holistic understanding includes renewable and non-renewable resources, although ecosystem services are of significant relative importance here. Reveal how near or far these two perspectives are when it comes to accounting ecosystem services, and more specifically those lacking market prices, is therefore the major underlying goal here.

At this point it is relevant to differentiate between green accounting and environmental accounting, as both concepts will appear throughout the rest of the dissertation. Green accounting is the terminology used in theoretical optimal control models when referring to the attempt of relating a comprehensive measure of NNP to the social welfare of a dynamic economy. An inclusive NNP includes more than the conventional market goods and services and net investment of man-made capital. When other ‘utilities’ related to the natural environment, as well as natural capital, are taken into account, the extension of the NNP is commonly referred to as GNNP (Aronsson and Löfgren, 1998a). Environmental accounting, by contrast, is not about measuring welfare, but economic activity. It is aimed at assessing, in monetary terms, the influence of natural resources into the national accounts, which are a set of macroeconomic accounts oriented towards providing a detailed picture of economic activities and the interactions between stakeholders. Hence, it obtains information on a wide range of natural resources and their use. At national level, it falls within the scope of the System of Environmental-Economic Accounting (SEEA). It is referred to as a ‘satellite’ accounting framework of the SNA, which we will speak about below. Thus, when mentioning conventional environmental accounting, we are talking about the role adopted in these differentiated accounting systems by the environment. Meanwhile, ecosystem accounting (or natural capital accounting) is no more than a subfield within environmental accounting. Consequently, it provides information on ecosystem stocks and flows so that it is possible to measure the ecosystem asset (natural capital)[8].

The chapter is organized as follows. We will start focusing primarily on environmental accounting. This section is divided into five subsections: (i) the first discusses conventional national accounting systems; (ii) the second analyzes the 'satellite' systems of accounts for natural resources and ecosystem services; (iii) the third assesses the valuation principles set out by these systems; (iv) the fourth describes an alternative accounting system that represents a considerable improvement on the current systems; (v) and the fifth presents various research projects aimed to estimate ecosystem services from different perspectives. Section two is dedicated to non-market valuation techniques. It is a different discipline based on obtaining monetary values for non-market resources. It is, however, more in line with environmental accounting than with the theoretical models that follow. Green accounting is left to the end, as the third chapter of this dissertation is closely related to the models presented here. This section begins with a description of the basic model pioneered by Weitzman in 1976, and continues with the extensions of this approach to include natural resources. It concludes with a brief explanation of the points of criticism made by alternative proposals to the original Weitzman's model.

2.1. Environmental accounting

Environmental accounts are relevant, as they permit, on the one hand, analyzing the interactions between the economy and the natural environment, and, on the other hand, knowing the state of natural stocks in terms of sustainability (Nordhaus and Kokkelenberg, 1999).

While the SNA is the international accounting standard, developed and developing countries, as well as supranational institutions, have made important advances with the

[8] It is playing a relevant role in Post-2015 development agenda and the Sustainable Development Goals. This may introduce the necessary conditions for a more sustainable development. Furthermore, in the near future, it may contribute to developing a new methodology including private and public income in ecosystem accounting, regardless of whether or not they have observable market prices (Campos, 2015).

view to helping this system become a more effective tool in policy making (Bos, 2003; OECD, 2004). Environmental accounts must play an essential role in this regard. Crucial for a sustainable development is indeed the integration of environmental policies into other policies[9] (European Council, 1999). In order to serve as a tool for such integrated policies, environmental accounts must be harmonized with national accounting standards.

In this respect, the United Nations Statistical Division (UNSD) promoted the creation of the Integrated Environmental and Economic Accounting in 1993 (later, in 2003, the SEEA) as 'satellite accounts' of the SNA. However, being under the influence of the SNA, the SEEA did not go beyond the market production, this being a major constraint. Ecosystems have been recently incorporated within the scope of the SEEA, giving rise to the SEEA-Experimental Ecosystem Accounting (SEEA-EEA) framework.

2.1.1. Conventional national accounting systems

The first SNA was presented in 1968 and remained valid until 1993, when the new version was endorsed. It was "a comprehensive and detailed framework for the systematic and integrated recording of the flows and stocks of an economy" (Bartelmus, 1989: 81). Nevertheless, although the SNA noted that man-made capital stocks must be included, which is right, it did not provide an explicit and comprehensive capital account. In fact, as the SNA primarily aimed to make recommendations on how to compile measures of economic activity, it was limited itself to the measurement of what is known as Net Valued Added (NVA)[10] or NDP, as it will be explained below.

Regarding the role of the environment for economic activity, the 1968 SNA suffered a number of shortcomings: only final consumption was incorporated and newly discovered reserves and changes in the values of reserves due to price changes were not

[9] In other institutional settings, several conference and forums on sustainable development, climate change and the environment have also been held since the 1990s to seek to address the concerns raised on these issues. In 1992, Rio de Janeiro was the setting for the Earth Summit or the UN Conference for Environment and Development (UNCED) that led to the Rio Declaration on Environment and Development. The Convention on Biodiversity, the Framework Convention on Climate Change or Agenda 21 were also adopted. A decade later, the World Summit on Sustainable Development allowed renewing the commitment to the principles of the Rio Declaration and Agenda 21.

[10] It corresponds to the difference between the market value from sales of final extracted products and the amortization of man-made capital.

taken into account either (Harrison, 1989). Although advances were introduced in the 1993 SNA[11], it confined itself to estimating an incomplete and inconsistent concept of national income – NVA or NDP – rather than tending towards the measurement of Hicksian income[12]. It therefore failed to adjust accounting indicators by the use of productions in progress (as part of costs) and natural growth of the year (as part of final production) (Campos, 2015). Depreciation of natural capital and environmental degradation were excluded as well. Non-market *amenities* were not included either. As a result, the GDP and NDP estimated by the SNA are recognized as providing an incomplete picture of the economy. In view of the fact that economic activity involves multiple interactions with the environment, these measures do not accurately assess the contribution and effect of the environment (Harris and Fraser, 2002).

A new version of the SNA was published in 2008. It was just an update of the 1993 SNA, rather than a revision. Some methodological and conceptual improvements for measuring economic activities were, however, undertaken in order to reflect the changes occurring since the 1990s (UN et al., 2008: 581-601; Eurostat, 2014: 27-28). Still, it did not prevent the new version from being immune to the required changes needed to calculate Hicksian income.

In parallel, other accounting systems were also developed in Europe and in the United States. The methodology of the national accounts put forward by the United States, called the National Income and Product Accounts (NIPAs) and produced by the Bureau of Economic Analysis (BEA), has given up measuring national total income. That is, it provides estimates for the monetary value of income and output per year. As this methodology is under the umbrella of the SNA, it is just limited to estimating NVA. In Europe, the first European System of Integrated Economic Accounts – also known as ESA – was implemented in the mid-1970s; while its integration process with the international proposals was accelerated at the end of the 1980s. The ESA-95 emerged in this context, so it is fully consistent with the 1993 SNA, especially as regards the accounting rules. In any case, it also took into consideration some of the specific

[11] It was created under the joint work of Eurostat, the United Nations, the Organization for Economic Cooperation and Development (OECD), the International Monetary Fund (IMF) and the World Bank. Upon publication, a harmonization of the different existing accounting systems in the world took place.

[12] In fact, economists have been expressing the need for official statistics to advance on a better measurement of Hicksian income (Campos, 2015).

characteristics of the European Union. It was later revised under the scope of the 2008 SNA, giving rise to the ESA-2010. It presented relevant differences compared with the ESA-95, notably in relation to the conceptual framework (see, European Commission (2013) for a manual on the changes between ESA-95 and ESA-2010).

2.1.2. 'Satellite' systems of accounts

The strong dependence of certain developing countries on natural and energy resources, on the one hand, and the negative effects of environmental degradation, on the other hand, made it necessary to develop an international framework for the environment. The UN Statistical Office (UNSO) presented the framework for the development of environmental statistics in the 1980s. However, it was in the 1990s that the UNSD prepared the ground for the development of the upcoming Integrated Environmental and Economic Accounting (later SEEA), whose preliminary version was tested in Mexico and Papua New Guinea (Hamilton and Lutz, 1996). It was therefore a framework to compile economic statistics and environmental statistics. The SEEA was published in 2003 as a system of 'satellite accounts' of the SNA. Following the guidelines laid down by the SNA though, the SEEA was considered to be more comprehensive due to the inclusion of households' accounts and some natural resources such as mineral, energy or water. However, non-market inputs and outputs and the discovery of new resources remained unacknowledged (Nordhaus and Kokkelenberg, 1999), as commercial NVA was again the only concept measured by this system. Production in progress and natural growth of the year were also ignored. Only current production was calculated, irrespective of the renewable resources to which it is applied (Campos, 2015). In any case, much of the debate about extending market limits to incorporate other commodities was not wound up (Landerfeld and McCulla, 2000)[13]. With regard to environmental degradation, the SEEA 2003 did not make any clear recommendation. Indeed, it proposed various methods such as cost-based valuation and damage-based valuation methods (UN et al., 2003: 394-395).

The UNSO led the development of the SEEA-Central Framework (SEEA-CF) (UN et al., 2009, 2014), which was published in 2014. It represents the current agreement between statistical offices for elaborating an international standard that can be

[13] The SNA and the SEEA only include direct use values for market goods and services. Direct use values for non-market *amenities*, indirect use values, option and passive use values are, by contrast, excluded.

successful in implementing accounts for marketed environmental assets. It has resulted in considerable advances towards linking physical environmental values to the economic production. Subsystems of the SEEA have been elaborated on specific resources (see, for example, SEEA-Water and SEEA-Energy).

In the United States, the prototype satellite environmental account system - US Integrated Environmental and Economic Satellite Accounts (IEESA) - was drawn on the experience of the SEEA. It did not take into account the change in the value of environmental stocks, however. In Europe, the Methodology on the Economic Accounts for Agriculture and Forestry (EEA/EAF) (European Communities, 2000) was developed for measuring the NVA of the most characteristic elements from the European forests (Lange, 2004), whereas non-market values and the capital balance were excluded (Campos and Caparrós, 2006).

2.1.2.1. Ecosystem accounting

The limitations of the SEEA and the EEA/EAF in measuring the total income of an ecosystem have generated a controversial debate over the amendments being made to these frameworks to construct ecosystem accounts (Campos, 2015). On the one hand, the Statistical Office of the European Communities (EUROSTAT) began working on this issue, but it took a long time for the statistical office to publish the pilot study European Framework for Integrated Environmental and Economic Accounting for Forest (IEEAF) (Eurostat, 2002). Either way, experimental results of the IEEAF so far have not gone beyond the limits of the production function of the ESA-2010 and its satellite EEA/EAF; while it integrates variations in forest capital during the accounting period instead (Caparrós et al., 2003).

On the other hand, the UNSD, in cooperation with the European Commission, the World Bank and the OECD, have recently complemented the SEEA-CF by integrating ecosystem accounting (i.e. the accounting for flows and stocks of environmental assets[14]), giving rise to the SEEA-EEA framework (UN et al., 2014). It is not yet

[14] "In the SEEA Central Framework environmental assets are measured from the perspective of 'individual' environmental assets, such as timber resources, land, mineral and energy resources, and water resources". "In contrast, the SEEA-EEA measures environmental assets from the perspective of the ecosystems…Ecosystem assets are thus environmental assets seen from a systems perspective" (European Commission et al., 2013: para. 1.19 and 1.20).

however an official standard. It relates ecosystems to economic activities in both physical and economic terms and covers some of the associated services classified by the report published by the Millennium Ecosystem Assessment (MA) in 2005 (Millennium Ecosystem Assessment, 2005: 15), such as provisioning services, regulating services and cultural services[15]. Criticisms of the MA (2005) are often related to the methodological weaknesses in terms of double-counting (Boyd and Banzhaf, 2007). One way of reducing the possibility of double-counting consists in better distinguishing between intermediate and final ecosystem services. Accordingly, progress is being made in the SEEA-EEA with a view to redefine this differentiation (Obst, 2015: 44-55). Likewise, although this system has its roots in conventional and 'satellite' systems of accounts, it aims to shed light on the non-market activity related to ecosystems and to integrate this information with market related data (UN et al., 2013: 1). It also attempts to make visible some market environmental services that are not recorded by current measurements of conventional national accounts (Campos, 2015) and to differentiate ecosystem services and assets in terms of the actors involved, namely public and private sectors (UN et al., 2013: 113)[16].

As mentioned above, the SNA does not provide an explicit accounting for environmental stocks. This gives an incomplete assessment of economic activity. By contrast, the SEEA-CF measures the stocks of environmental assets and changes in these stocks. In any case, ecosystem accounting described in the SEEA-EEA provides additional perspectives on this measurement, while principles of valuation are aligned between the two systems (we shall return to this issue later). For example, in the SEEA-EEA, biodiversity is considered as a characteristic of ecosystems rather than as an

[15] MA (2005) also includes habitat/supporting services. However, the classification of ecosystem services of the MA (2005) has been recently revised. The European Environment agency proposed the Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2013). This classification integrates supporting services into regulating services precisely to avoid double-counting.

[16] The project *Compiling and Refining Environmental and Economic Accounts* (CREEA) is intended to compile and refine environmental and economic accounts for water, waste and forest sectors within the limits of the SEEA-CF, while it does not advance in methodologies to integrate the monetary valuation of non-market ecosystem goods and services. It also works on issues related to climate change, emission allowances and land use changes. An interesting part of this project is that two databases, EXIOPOL and FORWAST, have been set up. The former is probably the most extensive economic-environmental database at global level. It follows the principles of the SNA and the SEEA. The latter provides physical waste and material accounts for the European Union countries (http://forwast.brgm.fr/ 02/11/2016).

ecosystem service, so falling biodiversity corresponds to declining ecosystem condition (UN et al., 2013: 48).

2.1.3. Valuation rules and principles in conventional and 'satellite' accounts

Regardless of the accounting framework used, there is a growing recognition of the utmost importance of economic valuation of environmental assets. Policy makers and international institutions require more data to draw up strategies, programs and policies oriented toward protecting the environment and mitigating the environmental degradation process (see, e.g. The Economics of Ecosystems and Biodiversity (TEEB) and Reducing Emissions from Deforestation and Forest Degradation (REDD+) initiatives). Thus, for accounts compiled in accordance with the SNA and the SEEA-CF the question of valuation in monetary terms is central.

The SEEA-CF recommends that exchange values be used[17] whenever market prices are observable (UN et al., 2014). In cases where this does not occur (for example, for goods that are recollected free and without paying a price such as mushrooms), the use of market price equivalents (i.e. the use of prices for similar markets) is proposed as the first criterion. It is considered that they provide an approximation to market prices (UN, 2008: 51; UN et al., 2014: 33). But some goods and services have neither market prices nor similar markets. Thus, the fact of giving a monetary value to them is a challenging task. The first solution that immediately comes to mind is to use the consumer surplus calculated with non-market valuation techniques (Bateman et al., 2002, 2013). However, the central normative national accounts framework excludes welfare measures for both market and non-market goods and services[18]. As pointed out by United Nations et al. (2003: 407), "one problem with the use of contingent valuation to value environmental damage is that it gives an average willingness to pay figure which includes an element of consumer surplus of indeterminate amount. This poses a problem when using contingent valuation in the accounting context, since the national accounts exclude consumer surplus". As for theoretical studies, they state that using consumer surplus for measuring non-market *amenities* is correct as long as one also includes it for

[17] Hultkrantz (1992) and Kriström and Skånberg (2001) apply this valuation principle for non-timber products and lichens, respectively, in Sweden, and so do Campos and Caparrós (2006) for hunting, grazing resources and mushroom gathering.

[18] Nonetheless, there is no consensus about how exchange values should be obtained for them.

market goods and services. It is just that most theoretical papers take it out for market goods and services.

With regard to environmental assets[19], the use of market price observations, or, alternatively, the use of information from similar assets would be, ideally, the best options. But considering that many of them are not traded in the marketplace, difficulties when applying the market price principle appear. This is, for example, the case of mineral and energy resources, natural aquatic resources and natural timber resources (UN et al., 2014: 33). That is why accounting frameworks propose two approaches: (i) written-down replacement costs, and (ii) Net Present Value (NPV). The former entails that the value of the asset will decline over time, as the acquisition price is reduced by the consumption of fixed capital over the assets life (UN. et al., 2014, 151). The second approach calculates the discounted present value of expected future returns (UN, 1993; UN et al., 2013, 2014)[20]. As the SEEA defines returns using the concept of economic rent[21], it may also be understood as the net present value of the economic rent to be generated for each of the future years. One needs first to estimate the current level of the resource rent, and then to make projections into the future. Future rents must finally be discounted to a current value (UN et al., 2003: 317). The NPV framework is also used for deriving valuations of flow measures of depletion, income and revaluation (UN et al., 2014: 219) and for investment decision-making. In the third chapter of this dissertation we will deal with this issue more extensively.

Beyond these accounting systems, yet related, there is also a line of research that seeks to estimate the real contribution of the forestry sector to the income generated by a country in an applied framework (Repetto et al. (1989) pioneered this approach for Indonesia's forests). Unfortunately, some studies have limited themselves to measuring total economic values, which include consumer surplus. In other words, they add exchange values for market goods and services and consumer surplus for non-market goods and services (the most well-known of these applications can probably be found in

[19] The same explanation applies to the stock of ecosystems – *ecosystem assets* –.

[20] It is in fact a dynamic CBA. It can either be done in terms of consumer surplus, and this is what most studies do, or in terms of market prices, as we do. In all cases, discounting the two measures is correct.

[21] It is defined as "the surplus value accruing to the extractor or user of an asset calculated after all costs and normal returns have been taken into account" (UN et al., 2014: 140).

Costanza et al. (1997)[22]). Thus, income is not estimated in a consistent manner with the criterion using exchange values (Fisher et al., 2008). Only a few papers have been following this practice (Caparrós et al., 2003, 2015; Campos and Caparrós, 2006; Obst et al., 2013[23]; Edens and Hein, 2013; Remme et al., 2015).

2.1.4. The Agroforestry Accounting System

The Agroforestry Accounting System (AAS) developed by the Group of Environmental Economics of the Consejo Superior de Investigaciones Científicas is a methodological proposal for ecosystem accounting as an alternative to the SEEA-EEA. Unlike the accounting systems explained above, in this case the measurement of the total income of an agroforestry system complies with the concept of Hicksian income. Total income incorporates economic products and costs by activities and unique products (this system also differentiates between private and social[24] income and measures incomes omitted by the EEA/EAF). Goods and services lacking both observable market prices and prices for similar markets are empirically simulated by using the SEV method to which we shall return later. The estimation of total income requires the compliance of the double-entry bookkeeping principle and the integration of the change in capital value at the end of the year. This permits estimating total income as net value added plus capital gains of the current year (Campos, 2015).

The fact that the AAS aggregates (*simulated*) exchange values for non-market *amenities* and exchange values for market goods and services and extends conventional and 'satellite' systems of accounts to the valuation of capital gains makes it a possible alternative to the SEEA-EEA. Extending the concept of production beyond market commodities by using (*simulated*) market prices follows the criterion proposed by the

[22] This study first estimates the value of global ecosystems by including consumer surplus and then compares it with GNP at market prices.

[23] More recently, Obst et al. also argue in favor of exchange values: "Despite their common motivations, the approaches of Bateman et al. and the SEEA differ in the ways that they assign value to ecosystem services. Bateman et al. ground their analysis in welfare changes as a consequence of specific policy scenarios. The SEEA approach aims to record the "output" generated by ecosystems, given current uses of ecosystem capital; thus, monetary values represent exchange values consistent with the principles of national accounting" (Obst et al., 2013: 420). Bateman et al. is referred to Bateman et al. (2013).

[24] Net transfers (i.e. subsidies minus taxes on product) and new discoveries during the accounting period are not included.

international normative frameworks and guarantees the consistency when aggregating public and private incomes.

Furthermore, although the AAS is a system of microeconomic accounts applied in multiple-use forests (Caparrós et al., 2003; Campos and Caparrós, 2006; Campos et al., 2008; Campos, 2010; Oviedo et al., 2013), it can also be used as a whole-farm approach. The project *Renta y Capital de los Montes de Andalucía* (*RECAMAN)* has implemented this accounting system and estimated the geo-referenced values for a whole Spanish region, Andalucía (Caparrós et al., 2016).

2.1.5. Global, regional and national research projects

In this context, several research projects have been launched over the last years with the goal of estimating the value of one or various ecosystem goods and services in some parts of the world. The study of these projects is relevant, as it permits analyzing what is being done in applied research. Most of these studies follow a multidisciplinary approach and are not intended to draw up economic accounts for environmental assets. Their main interest lies in the empirical estimation of the variation of consumer surpluses derived from comparisons of simulated management scenarios. These studies therefore perform a CBA of large-scale assessments. The project *RECAMAN* is a remarkable exception (we shall return to that later). As explained in section 2.1.3, conventional and 'satellite' systems of account are very clear in rejecting the use of consumer surplus for measuring goods and services with neither market prices nor market price equivalents. Still, the explanation of these projects helps strengthen the understanding of what this dissertation proposes and conducts to estimate the value of ecosystem services in a consistent way with the principles of accounting systems.

In 2007, G8+5 Environment Ministers proposed a global initiative to analyze the value of biodiversity loss, ecosystem degradation, economic benefits of biological diversity and cost of conservation in such a way that informed decisions can be taken at all levels. The results of this study led to the initiative *The Economics of Ecosystems and Biodiversity (TEEB)*. It is inspired by the Stern Review on the Economics of Climate Change and the proposals of the Intergovernmental Panel on Climate Change (IPCC) (Ring et al., 2010). Some countries and international institutions, including the European Commission and the United Nations Environment Programme (UNEP), set in motion this initiative and put into practice political actions to move forward on the evaluation of future economic consequences of biodiversity loss. Although there is no a specific methodological proposal in relation to ecosystem accounts, its view on the valuation of ecosystem services is in favor of applying the TEV approach. It also recognizes the importance of non-market ecosystem services when designing public environmental policies (Campos, 2015). Study reports were accomplished in 2010, allowing several countries to join the initiative and to seek assistance to gather information to improve their own national systems (www.teebweb.org 02/11/2016).

In the same vein, the European Environment Agency has also been effective in developing *TEEB* by proposing the aforementioned *CICES*. Indeed, the European project *EURECA* was promoted by this very same institution in 2008 and oriented towards evaluating European ecosystems. It is one of the evaluations under the follow-on program of MA (2005) and was designed, in part, to bring together all national projects. Among them are differences over whether to use economic valuation for the development of ecosystem accounts (Campos, 2015).

One of the projects in favor of estimating economic values for non-market goods and services is *UK National Ecosystem Assessment (UK NEA)*. It is financed by the Department for Environment, Food and Rural Affairs of the United Kingdom and other institutions involving experts on natural, economic and social sciences. Although it is a successor to the MA (2005), it brings some conceptual and scientific advances. However, the main objective is to analyze and quantify the status of ecosystem processes and the services they provide across the United Kingdom (Bateman et al., 2011). It is, thus, more focused on non-market valuation, rather than on environmental accounting itself. Even so, it deserves to be mentioned, as in chapters four and five of this dissertation we also focus on issues related to non-market valuation techniques, while in a different sense. Going into more detail, this project promotes the valuation of ecosystem services from a welfare perspective derived from the modeling of simulated scenarios of future management of ecosystems (Bateman et al., 2011). Simulations from the present until 2060 to better understand the consequences of alternative land-use features are carried out (Bateman et al., 2013).

Renta y Capital de los Montes de Andalucía (RECAMAN) is one of the most ambitious projects ever to be undertaken in the field of environmental accounting. Unlike other research efforts, *RECAMAN* has been done within an accounting framework, rather than within CBA. The AAS methodology has been applied in order to measure total Hicksian income in multiple-use forests of this southern Spanish region. Goods and services for which income is calculated are: timber, cork, firewood, forest dried fruits, grazed fodder, conservation forestry, government forestry, forestry, game, residential services, private *amenities*, public recreation, mushrooms, carbon, landscape, biodiversity and forest water. This methodology has also been applied to a group of 58 estates with various cases-types of forest vegetation in an experimental pilot scheme. 33 of them are private estate; whereas 25 are public (Ovando et al., 2015). They have been later geo-referenced at regional scale, where the results for each main forest species present in Andalucía are presented. The aggregation by vegetation formations is also analyzed (Caparrós et al., 2016). The application of the AAS at regional scale may encourage the governments of the European Union to implement ecosystem accounting in the national accounts by 2020.

In parallel with the above, albeit from the SEEA perspective, the World Bank project launched in 2010 the project *Wealth Accounting and the Valuation of Ecosystem Services (WAVES).* Taking the SEEA-CF as the conceptual basis, it seeks to develop a methodology to estimate total income and total capital of ecosystems, including some non-market goods and services and more inputs than those currently covered (WAVES, 2013). Various pilot applications are being put in place in eight countries: Botswana, Colombia, Costa Rica, Guatemala, Indonesia, Madagascar, the Philippines and Rwanda (www.wavespartnership.org 02/11/2016). The Technical Experts Committee is made up of experts from different countries and institutions. This project also works closely with the UNSD to develop a tool to help guide countries apply the SEEA at national level.

2.2. Non-market valuation techniques

At this point, it is important to address the issue of non-market valuation methods, as most applied studies on non-market ecosystem services use them to calculate their economic value. We have just seen that these techniques have also been the line of work followed by research projects such as the *UK NEA*. In chapters four and five, we also deal with these techniques, although from a different perspective to that used in most of these studies. While non-market valuation techniques are generally required for use in CBA, in these two chapters they are utilized to obtain the (*simulated*) exchange value of public forest recreation and biodiversity conservation in Spain.

As explained in the introductory part, the main purpose of CBA is to estimate the costs and benefits that individuals may experience resulting from a change (positive or negative) in an ecosystem good or service. The basic concepts that are used to measure such costs and benefits are WTP and WTA. Non-market valuation techniques calculate these two economic values, thereby deriving consumer surplus or Hicksian variation measures. Aggregating individuals' preferences gives the welfare value that the whole society attaches to the change in the ecosystem good or service under consideration.

Preferences can be valued as a result of non-market behavior (actual or intended) according to their own characteristics. Revealed preferences (RP) techniques analyze actual behavior; while stated preference (SP) techniques assess intended behavior by constructing hypothetical markets that elicit individual preferences for a change in the provision of a good or service (Atkinson, 2010). Given that the fourth and fifth chapters of this dissertation deal with SP techniques, we will focus on these techniques.

SP methods have received growing attention over the last decades owing to its flexibility and ability to estimate passive use values (Atkinson and Mourato, 2008; Hoyos et al., 2012). The economic value of non-market ecosystem services is obtained by asking people directly, via questionnaires, how much they are willing to pay to change the condition of the good or service in question or to preserve it, rather than by looking at its influence on actual markets for some other goods or services (Bateman et al., 2002) [25]. Hypothetical markets are used to analyze intended behavior, thus allowing ex-ante judgments to be made (Mitchell and Carson, 1989: 89). Applying SP techniques therefore involve taking into account behavioral patterns, as well as the heterogeneity of preferences within or between individuals (Farizo et al., 2014a, 2016) [26]. Out of these techniques, contingent valuation and choice modeling are the most important. Both methods share the theoretical framework in the random utility approach developed by McFadden in 1974 (Hanley et al., 2007: 341).

[25] Respondents may also be asked for the WTA for a loss, but this format is not very often in SP studies (Mitchell and Carson, 1989: 2).

[26] Traditionally, heterogeneity has been included through variables reflecting individual characteristics such as education, gender, age or income, so the effect of these variables on individuals' utility and WTP can be explored. Attitudinal and behavioral factors, as well as the influence of the socio-cultural and regional context in which individuals live, have often been disregarded. However, their inclusion in SP methods has been increasing over the last years (Boxall and Adamowicz, 2002; Johnston, 2007; Soliño et al., 2009; Farizo et al., 2014a,b, 2016; Hoyos et al., 2015).

Contingent valuation has been extensively applied over the last decades and a large body of empirical and methodological research has developed (Nyborg, 1996; Hanley et al., 2007). Using an appropriately designed questionnaire[27], the hypothetical market is described. There are several variants in the formulation of the question aimed at obtaining a price for the good or service (Riera, 1994). Questionnaire usually starts asking whether he/she is willing to pay a certain amount of money. If so, question may be repeated increasing the amount. If not, a lower amount is offered (Hanemann and Kannninen, 1999). Finally, it is often asked what the maximum price that he/she would be willing to pay would be, considering previous responses. Questionnaire survey may therefore be conducted using closed-ended formats and open-ended formats. The main difference between the situation faced by the respondent in a contingent valuation scenario or in a real market is that in the former case the market is hypothetical so that he/she does not have to pay the stated price (see, Mitchell and Carson (1989); Hausman (1993); Bjornstad and Kahn (1996); Bateman et al. (2002); Champ et al. (2002); Alberini and Kahn (2006) for a thorough review). In the fifth chapter we conduct a contingent valuation study to calculate the demand function for a conservation program to protect the Iberian Lynx in Spain.

Choice modeling (Louviere and Hensher, 1982; Louviere and Woodworth, 1983; Louviere et al., 2000) offers respondents to choose among different alternatives described by a set of attributes, including a payment vehicle, such as a fee, a contribution to a fund or a tax increase, that they would have to pay. As respondents may evaluate attributes separately, choice modeling is more appropriate for valuing specific attributes (Hanley et al., 1998a,b; Morrison et al., 2002); while contingent valuation is better suited to analyze the general result of a change in the non-market ecosystem service (Baker and Ruting, 2014). Choice modeling uses a range of formats, including rating, ranking[28] and choice. The last two formats (although ranking in a lesser degree) tend to be preferred, as they are more readily interpreted in terms of random utility (Roe et al., 1996). In choice experiments respondents are requested to

[27] The theoretical proposal of using a survey method was first introduced by Ciriacy-Wantrup in 1951. It was not, however, until 1963 when R. David empirically implemented the contingent valuation method for the first time.

[28] They are also known as contingent rankings (Laureu and Rae, 1989; Hanley et al., 2001; Foster and Mourato, 2000; 2002; Bateman et al., 2006; Varela et al., 2013).

choose one alternative out of a set[29]; while in rankings they are requested to rank the options offered (see, Hoyos (2012)). Choice experiments are increasingly used in research, as they estimate the value of non-market goods and services by making separate assessment of the respondents' preferences for the attributes of the good or service. In the fourth chapter, we use a choice experiment to estimate the demand function for public recreation in stone pine and cork oak forests in Spain.

For many years, contingent valuation was not very popular amongst researchers, following the publication of a critical paper by Samuelson in 1954. Being public goods, individuals who do not pay anything cannot be excluded from consuming them. Thus, respondents may provide a strategic response by indicating a different price than the one they are actually willing to pay. This strategic bias may be associated with a free-rider problem that led Samuelson to discourage this method (Riera, 1994; Jakobsson and Dragun, 1996: 83; Champ, 2002; Carson and Hanemann, 2005; Carson, 2012).

In the same vein, the validity of the contingent valuation method (and of stated preference techniques in general) has also often been called into question (Hausman, 2012) due to their hypothetical nature which makes difficult to discern if stated choices would be actual choices in real markets. As far as back as the 1970s, relevant improvements were made to the reliability of the method (Bohm, 1972; Randall et al., 1974; Brookshire et al., 1976; Rowe et al., 1980; Schulze et al., 1981). These advances boosted the popularity of contingent valuation. But it was in the 80's when this technique gained ground thanks to the works of Cummings et al. (1986) and Mitchell and Carson (1989). Both studies brought together practitioners, economists and psychologists with the goal of placing this technique in a broader and more multidisciplinary context than that of environmental economics (Hanemann, 1994; Riera, 1994). In 1989 the Exxon Valdez oil spill occurred. Government agencies realized that the compensation of damaged included passive use values. In fact, one clear advantage of this method, and of SP techniques in general, is that, unlike RP techniques, they can be used to estimate passive use values, which are likely to be important when it comes to non-market ecosystem services such as biodiversity conservation. In order to evaluate economically the damaged caused by the oil spill, contingent valuation method was therefore proposed. While objections were raised, the

[29] Accordingly, contingent valuation can be seen as a subtype within choice modeling.

NOAA panel's report (1993) rejected the criticisms expressed and ruled in favor of the use of this method. Since then, contingent valuation, but also choice modeling, has become increasingly widespread amongst researchers.

Concerning the aspect of the hypothetical nature of these techniques, there is a one way to validate them. It consists in testing their internal consistency by targeting the key elements in the valuation scenarios that guarantee a realistic market simulation. This means testing for construct validity, which includes theoretical and convergent validity (Whitehead et al., 1995; Whitehead et al., 1998). Theoretical validity (also termed internal validity) refers to extent to which the findings of the study are consistent with theoretical predictions and *a priori* expectations (Schläpfer, 2008)[30]; while convergent validity assesses whether two measures of WTP obtained using different methods or choice techniques are correlated and therefore converge to similar estimates (Diamond and Hausman, 1993).

2.3. The Simulated Exchange Value method

The SEV method estimates exchange values for ecosystem goods and services that are currently outside the market, i.e. for which there are neither market prices nor similar markets. This estimation helps ensure consistency with market based figures considered in the national accounts.

As explained above, normative frameworks recommend that exchange values be used as long as market prices are observable in the marketplace. Hence the SEV method proposes to simulate, in a partial equilibrium model, the (*simulated*) market price that one could get if non-market ecosystem services, such as public forest (free access) recreation, were internalized (Caparrós et al., 2003, 2015). This differs from CBA which uses different Hicksian variations, namely welfare measures for non-market ecosystem services (Bateman et al., 2003).

[30] One way of testing it is to analyze the determinants of WTP (Mitchell and Carson, 1989: 206) such as income (e.g. the higher the income, the greater the probability of paying). Interesting papers analyzing internal validity of choice experiments can be found in Soliño et al. (2012, 2016). The former is aimed at estimating consumers' preferences for a policy of replacing conventional electricity sources with electricity generated by forest biomass, while the latter analyzes the preferences of Andalusian hunters for different driven hunts.

The SEV method simulates the whole market (demand, offer and competitive environment). The only way of internalizing consumer surplus in a market is to assume that each individual would pay their maximum WTP, namely a differentiated price to each individual. As his assumption is far-fetched, the premise of the SEV method is that there would be a single price for the provision of the service with the goal of maximizing profits. To get this price, this method uses a demand function (a WTP function) estimated with one of the non-market valuation techniques previously explained, and a supply function based on the commercial costs associated with the provision of the service. Demand functions can be simulated using the travel cost method, thus obtaining the marshallian demand function, or alternatively using SP methods. In the latter case the Hicksian demand function is obtained, so the marshallian demand can be calculated under the assumption that the wealth effect is of small significance.

Having discussed the procedure to determine the price, the next issue consists in estimating the number of units consumed at that price. The common procedure followed by studies estimating ecosystem values based on prices for similar markets involves multiplying the price by all the units consumed outside the market (see, e.g. Hultkrantz (1992); Kriström and Skånberg (2001)). This is the proposal of the SNA. According to this approach, setting a price would not result in a drop in the number of units consumed, which does not make sense. However, from the demand functions we know that when a price is set, only part of the population would pay that price in the case that the non-market services were internalized. For example, if a price is equal to the estimated value of the median of the WTP of a sample of visitors to a forest area, only 50% of them would be willing to pay this price (Caparrós et al., 2003). The applied studies conducted in the fourth and fifth chapters of this dissertation show that this percentage is not exactly 50 per cent, but close to it. As we will see, under the SEV scenario, revenue function is defined as the (*simulated*) price multiplied by the number of individuals that would consume the service (in the case of public forest recreation, we would refer to the number of visitors) in the hypothetical market. That is, the SEV method only considers the number of individuals that would accept a payment.

Advantages of this method include the possibility of making international comparisons. While some countries, such as Spain, do not usually charge any price for forest recreation, other countries do so. However, it suffers from the problem associated with all the extensions of the national accounts based on simulations (Caparrós et al., 2015). Actually, the use of the SEV is an interesting proposal that may be considered an important improvement compared with omitting values for non-market ecosystem services or with estimating welfare measures within a CBA analysis. This means that associating environmental accounting with the calculation of the TEV is not correct in that it supposes the estimation of consumer surplus or any other Hicksian variation. By contrast, the huge advantage of the SEV method is that it permits linking environmental accounting and green accounting in the same light. This method is therefore the backbone of the dissertation, as will become apparent in subsequent chapters.

2.4. Green accounting

This section approaches the incorporation of natural resources into the economic analysis from the perspective of national income accounting. Consistent with this, the issue analyzed is whether NNP is the appropriate indicator of economic performance and social welfare. This in turn raises the question of whether NNP growth entails a welfare improvement over time. It is not less important to look at whether answers to these questions remain valid when it comes to GNNP. Although natural resources had generally been excluded from the standard neoclassical growth model (Nordhaus and Tobin, 1972), there has been a commitment to greening national accounting since the beginning of the 1990s. Interest in the relationship between income, wealth and social well-being has indeed emerged in recent decades due to the growing concern about the long-term negative consequences of environmental degradation and natural resource depletion (Ferreira and Vincent, 2005).

The main challenge facing green national accounting is, therefore, how to make it more comprehensive so that depletion of natural resources (Pemberton and Ulph, 2001) and goods and services with and without market prices may effectively be assessed. However, ‘greening’ national accounting is by no means straightforward, as it involves regarding different perspectives and purposes.

In this section, we critically analyze the progress made in this field over the past decades. We start from the welfare interpretation of NNP given by Weitzman in his seminal 1976 paper. This requires considering some of the underlying assumptions on what this statement is based. Weitzman's model has been extended to cover natural resources (green national accounting), especially exhaustible and renewable resources.

As this dissertation focuses on forests, most of this section is dedicated to analyze and explain theoretical frameworks on forests. This will help follow the steps given in the model set out in the third chapter.

2.4.1. National Income, Net National Product and Social Welfare

Drawing on the definitions of income proposed by Hicks and Fisher and already analyzed in the introductory chapter, national accounting is mainly targeted to answering whether a country will be as well off in the future as in the present. Although apparently general, this question is neither simple nor a foregone conclusion. It includes not only measuring current and future welfare, but also knowing whether the actual development is sustainable. This also leads to the question of which measures may be used in order to achieve these purposes.

Measuring social welfare involves measuring individuals' utility, which is a function of consumption. As utility cannot be directly observed and it is just a theoretical construct without objective measurement unit (Li and Löfgren, 2002), a measurable indicator of social welfare is needed. The standard measure of current welfare in the national accounts is the aggregate market value of consumption. However, one also needs another summary consumption-based measure of future welfare (Vincent, 2000) in order to allow for a direct comparison. The question has arisen as to whether NNP may be interpreted as the suitable indicator in this respect. This has resulted in an intense debate over the last decades. Contrary to the idea supported by earlier economists including Fisher, Samuelson (1961) argues that current national income concepts must be rejected if a welfare concept is to be searched for and proposes to use wealth-like magnitudes[31], such as the present discounted value of future consumption. Formally, one may construct what is known as 'wealth-equivalent income' (or Hicksian income): the consumption that if held constant will yields the same wealth – or, alternatively, the same discounted sum of consumption – as wealth-maximizing path (Asheim, 2000).

Instead, Weitzman (1976) demonstrates that, under certain special assumptions, conventionally defined NNP includes not only current consumption, but also the present

[31] "The only valid approximation to a measure of welfare comes from computing *wealth-like* magnitudes, not income magnitudes of the Haig, Fisher or any other type" (Samuelson, 1961: 57).

discounted value of future consumption[32], thus bridging the gap between Samuelson's statement and current income measures. These assumptions are: a single consumption good, linear utility, stationary technology (i.e. technology is independent of time), and constant interest rate (or utility discount rate). In the optimal control model developed in the third chapter of this dissertation we also take into account these assumptions.

On this basis, the initial question may now be worded as follows: is the economy able to keep consumption constant over time such that future generations may have the possibility of consuming at the same level as current generations?

Weitzman (1976) is inclined to answer the question in the affirmative. Let us see the argument: Imagine that economy consists of a single consumption good and n capital goods (including pools of exhaustible resources). Consumption level at time t is denoted $C(t)$, the stock of capital is $K(t)$ and net investment flow is $i(t) = dK(t)/dt$. However, as consumption is the ultimate end of economic activity, investment is an intermediate good whose final products are the consumptions of different years.

The aim of the Social Planner is to maximize households' utility (social welfare)

$$\max \int_0^\infty C(t) e^{-rt} dt \qquad [2.1]$$

subject to the resource constraint of the economy and an initial condition for the capital stock:

$$\left(C(t), i(t)\right) \in \Phi\left(K(t)\right) \qquad [2.2]$$

$$K(0) = K_0 \qquad [2.3]$$

where $\Phi(K)$ is the production possibilities set.

The necessary conditions for any solution of this problem are:

$$Y\left(K^*(t), p_i(t)\right) = C^*(t) + p_i(t)\frac{dK^*}{dt}(t) \qquad [2.4]$$

[32] "It is not really a question of choosing between a conventional but inappropriate current income concept and an impractical but correct wealth-like magnitude, because in principle they are merely different sides of the same coin […] the welfare justification of net national product is just the idea that in theory it is a proxy for the present discounted value of future consumption" (Weitzman, 1976: 156).

$$\frac{dY}{dK_i^*}(t) = rp_i(t) - \frac{dp_i}{dt}(t) \qquad i = 1,\ldots, n. \qquad [2.5]$$

where [2.4] represents NNP (with consumption as numeraire) along the competitive trajectory. From this equation is also derived that relative prices equal marginal rate of transformation. Equation [2.5] is the intertemporal efficiency condition of a competitive capital market, being p_i the cost of buying an extra unit of capital good *i* (see footnote 5 in Weitzman (1976) for a further explanation) and *r* the rate of return.

In order to be able to provide a welfare interpretation of NNP, understanding this to be the largest permanently maintainable value of consumption, investment goods must be weighted with prices measuring marginal rate of transformation, as in equation [2.4]. The main advantage is that, if all investment may effectively be convertible into consumption at the given price-transformation rate, NNP as currently defined in [2.4] may be expressed as "the maximum attainable level of consumption that could be maintained forever without running down capital stock" (Weitzman, 1976: 159). That is, NNP would be equivalent to what is known as 'welfare-equivalent income': the consumption that if held constant will yield the same welfare – i.e. the same discounted sum of utility – as the welfare-maximizing path (Asheim, 2000; Vincent, 2000).

However, considering that marginal transformation rates cannot be generally used to transform investment goods into consumption, the consumption level in [2.4] is not attainable at time *t* [33]. In other words, it is based on a hypothetical consumption path. Hence, if one wants to estimate the maximum welfare actually attainable, it is required to measure what is known as sustainable income [34]: the maximum sustainable consumption – i.e. the maximum consumption that can actually be sustained over time – (Cairns, 2000; Pezzey and Toman, 2002). Weitzman (1976) demonstrates that:

[33] The first definition of stationary welfare equivalence as the *hypothetical* constant utility level that would yield exactly the same welfare as the welfare-maximizing path can be found in Weitzman (1970).

[34] Sustainable income is related to the term sustainable development. While sustainability (Pezzey, 1989, 1992; Hanley, 2000; Hamilton, 2000) has been the center of many debates among environmental economists, this concept was not defined until 1987 when the report "Our Common Future" was made public. It follows that if current consumption exceeds sustainable income, sustainability is not achieved, as consumption would need to be lower in the future than at the present time (Vincent, 2000). Thus, sustainability will only be secured if $U(s) \geq 0$ for all $s \geq t$ (Pezzey, 1997; Asheim, 2000).

$$\int_t^{\infty} C^*(s)e^{-r(s-t)}ds = \int_t^{\infty}\left[C^*(t) + p_i(t)\frac{dK^*}{dt}(t)\right]e^{-r(s-t)}ds \quad [2.6]$$

where sustainable income (the R.H.S.) is equivalent as the maximum welfare actually attainable (the L.H.S), which, in turn, is the same as what would be obtained from the hypothetical constant consumption level.

Weitzman’s result therefore confirms the definition of NNP as the stationary equivalent of future consumption. This provides a measure of social welfare and gives economic form to the concept of sustainability. Moreover, as utility function is assumed to be linear – i.e. $U(C(t)) = C(t)$ –, current utility equals the market value of consumption, so wealth-equivalent income and NNP are equivalent to each other and with welfare-equivalent income, thereby reconciling Fisher-Lindahl-Hicks' view with optimal growth theory (Aronsson and Löfgren, 1998a; Asheim and Buchholz, 2004; Asheim, 2007).

In addition to the foregoing, the most striking result derived from the seminal paper by Weitzman is that the Hamiltonian for a general optimization problem of the form [2.1] – [2.3] can be interpreted, under the above-mentioned assumptions, as being equivalent to the NNP measured by the national accounts (see, Asheim (2003) for a taxonomy of different results, and the related assumptions). The Hamiltonian is a function that summarizes the objective function and the constraints of an intertemporal optimization problem by using only values at the current instant (the current year in the case of NNP). It suffices for it to be maximized in order to achieve the optimal path. All the same, as explained, these results can only be achieved due to the assumptions made. As we will see in the final part of the chapter, a strand of literature exploring with a critical view some of these assumptions has been developed.

The basic result by Weitzman (1976) has been later extended, but it has shown that it continues to be valid, provided that the assumptions made are met. The most advanced model so far can be found in Sefton and Weale (2006). After taking into account the criticisms of the Weitzman’s work over the last decades, they obtain similar results. They analyze real NNP as a measure of social welfare from a decentralized economy, defining NNP in terms of consumption from the production side and relating net savings (real income minus consumption) to the rate of change of households’ well-being from the consumption side[35]. NNP is thus equivalent to the Hamiltonian and to the national income, the latter being understood as wealth (return of capital) (Cairns, 2008). Yet this is only true in case of non-convexities (Cairns, 2003). On this basis, Asheim and Wei (2009) develop the results achieved by Sefton and Weale (2006) for the case when the interest is on sectorial incomes.

Still, on the basis of this approach, national accounting has been expanding the concept of NNP to include the value of environment in terms of welfare (Cairns, 2002). Many efforts have been made since the beginning of the 1990s to incorporate exhaustible resources, renewable resources and the negative effects of pollution. Logically, as the title indicates and in order to remain consistent with the main objective of the dissertation, we will devote most of this section to explaining theoretical models focused on goods and services provided by forests.

2.4.2. Exhaustible resources

The consumption of a resource unit involves its complete destruction and its regeneration requires a very long period of time. Exhaustible resources are used up in the production process and their growth rates are zero (Dasgupta and Heal, 1979: 153), so the intertemporal sum of the services they provide is limited.

Hotelling (1931) did open up the way for a new literature by turning his attention to the appropriate time of extraction and the rate at which resource deposits should be extracted (Turner et al., 1994). Hotelling's rule states that the present value of a homogeneous exhaustible resource should be identical regardless of its extraction time. It turns out that the price of these resources should grow at a rate equal to the interest rate in an efficient and competitive economy. The same conclusion is shared by Stiglitz (1976) in the case of infinite time horizon and absence of extraction costs[36].

[35] The relation between real savings and the change in social welfare is grounded in the Hartwick rule: consumption will be constant over time if the value of net investment is zero (Hartwick, 1977). This is equivalent to making the present value of future changes consumption zero.

[36] Romero (2012) derives an interesting theoretical framework in which the maximization of the NPV associated with the exploitation of a given resource is given by the sum of the present value of the amount of resource extracted, the present value of the accumulated flow of possible non-market services provided by the resource across its planning horizon and the present value of the accumulated flow of possible land

The Dasgupta-Heal-Solow-Stiglitz (DHSS) model (Solow, 1974; Dasgupta and Heal, 1974; Stiglitz, 1974) assumes that the stock of exhaustible resources is an input of production that can either be consumed or used for net investment in man-made capital (Benchekroun and Withagen, 2011). It helps analyze whether accumulation in augmentable capital may compensate their depletion (Mitra et al., 2012). This is the same to the Hartwick rule: if society invests rents from exhaustible resources and profits from instruments of wealth in reproducible capital, consumption and output will remain constant (Hartwick, 1977; Hanley et al., 2014). This means that a specific investment policy should be designed in order to achieve a constant consumption path.

From a green accounting point of view, and on the basis of the Weitzman's (1976) approach, Solow (1986) inquires about the intertemporal allocation of non-renewable resources and gives a growth-theoretic interpretation of the Hatwick rule[37]. On the one hand, the conventional Ramsey optimal growth and capital accumulation model takes a utilitarian approach. It assumes that the sum of instantaneous utilities of consumption over time represents a measure of social welfare and that each generation's utility only depends on its own consumption. Thus, it is possible to bring about equity between generations by adding the utility level of different generations[38]. On the other hand, intergenerational equity has also been analyzed from the Rawlsian perspective: "the welfare criterion is the standard of consumption of the least well-off generation" (Solow, 1986: 143; Swenson, 1986: 1). However, Solow (1986) argues that these two approaches have many problems. Thus, he regards consumption as the interest on the stock of the non-renewable resource. Applying the Hartwick Rule involves that

rents that the use of the resource entails. By calculating the first order conditions, the basic rule for the optimal economic exploitation is found.

[37] The absence of exogenous technological progress implies that, if consumption is constant at all future time, the value of net investment is zero at time *t*.

[38] The key question is to analyze under which conditions sustainability is consistent with optimal growth. This is based on the Golden Rule of economic growth, i.e. the growth path that gives the largest level per capita consumption that be maintained permanently. This utility-based definition of sustainability differentiates between weak and strong sustainability (see, Garmendia et al. (2010) for a comparison on fisheries management). Given the existence of a critical natural capital for which no substitutes exist (strong sustainability), weak sustainability assumes that there are substitutes for all assets and that only the maintenance of total capital stocks is relevant to generate future utility (Solow, 1974; Hartwick, 1977; Pearce and Atkinson, 1993; Neumayer, 2003). Moreover, from equation [2.5] it is also shown that weak sustainability and intertemporal efficiency are not conflicting (Olschewsky and Klein, 2011).

$\dot{K}(t) = 0$ in [2.4], while the reproducible capital goods are included as net investment in the conventional way. In the case of the exhaustible resource, $-\dot{K}(t)$ is the current rate of depletion and $-\dot{K}(t)p_i(t)$ is the market rent when there are no extracting costs. This means that $\dot{K}(t)p_i(t) = 0$, so the sum of current resource rentals equals current net investment in reproducible capital.

Hartwick (1990) goes beyond and states that changes in the stock of exhaustible resources are defined as the difference between the stock of new discoveries and the current flow. There are also costs of extraction and exploration. By linking linearized current-value Hamiltonian to GNNP, it is shown that current 'Hotelling Rents' on net stock diminution over the period (i.e. the difference between marginal product and marginal cost of extraction multiplied by the variation in the stock) should be deducted from green GNP to arrive at GNNP.

Cairns' (2002) model is in line with the previous approach, albeit more complex. The economy has a stock of exhaustible resources and it is subject to pollution, so the arguments of the utility function are consumption of composite goods, the stock of pollution and the flow of environmental *amenities*, which is equal to the negative change in the stock of resources. It is also assumed that the utility flow is discounted at a rate $\alpha(t)$, so the resulting maximization problem is exactly what CBA would pursue. However, Cairns' approach differs from CBA, as it takes out consumer surplus, which allows him to relate the linearized current-value Hamiltonian to GNNP.

2.4.3. Renewable resources

These natural resources can be used repeatedly, as they regenerate themselves naturally. This entails that they are practically inexhaustible. Solar energy, fisheries and wood are the clearest examples.

The first approach including renewable resources within a green accounting framework can be found in Hartwick (1990). This model is relevant for the objective of this dissertation, as, while focused on fisheries, it opened up the possibility of developing other similar studies on forests, as will be discussed later.

Utility is derived from current aggregate consumption of a composite good and fish A with linear approximation, $U(C)C + U(A)A$ [39]. Total production of the economy, which is function of the stock of man-made capital and current labor force, ℓ , is distributed between consumption, investment and fishing costs $\xi(A, R)$, being R the stock of fish and h the current harvest.

The Social Planner's problem is (*r* is the discount rate):

$$\max \int_t^{\infty} U(C, A) e^{-r(s-t)} ds \qquad [2.7]$$

subject to:

$$\dot{K} = Y(K, \ell) - C - \xi(A, R) \qquad [2.8]$$

$$\dot{R} = g(R) - h \qquad [2.9]$$

where $g(R)$ is the natural growth in the stock.

The current-value Hamiltonian for this problem (in utility terms) can be written as:

$$H_c = U(C, A) + \lambda_1 [Y(K, \ell) - C - \xi(A, R)] + \lambda_2 [g(R) - h] \qquad [2.10]$$

Using the linear approximation above-mentioned and dividing [2.10] by the utility, the current-value Hamiltonian can be rewritten in 'monetary' values. At the same time, by calculating the canonical equations, NNP can be derived as:

$$NNP = \frac{H_c}{U(C)} = C + A \frac{U(A)}{U(C)} + \dot{K} + \left[\frac{U(A)}{U(C)} - \xi_A \right] \dot{R} \qquad [2.11]$$

where $\frac{U(A)}{U(C)}$ is the market price of a unit of fish and ξ_A the marginal cost of fishing. The last term can be considered as economic depreciation when $\dot{R} < 0$ [40]. By deducting this

[39] As the models set out in this section are only derived at time *t*, we take out any source of time-dependence for simplicity.

[40] As the existence of clear national property rights make distortions in prices and quantities smaller, it is better to define depreciation in terms of current harvests and natural growth. However, for certain renewable resources such as fish there is a difficulty in establishing property rights.

term from GNP, one calculates NNP. This approach thus shows that the current-value Hamiltonian is equivalent to NNP.

Following from this work, two types of alternative approaches (which may, however, be used in combination) may be drawn up for dealing with goods and services provided by forests, as well as with the interactions between the forestry sector and the rest of economic sectors: noteworthy are the studies by Cairns (2001, 2003), on the one hand, and by Vincent (1999a) and Matero and Saastamoinen (2007), on the other hand.

A similar model as the one of Hartwick (1990) can be found in Cairns (2001), though focused on forests. The Social Planner seeks to maximize households' utility (social welfare), which is generated by consumption of a market good and forest *amenities* $A(R)$ which depend on the stock of environmental asset. Unlike in Hartwick (1990), resource constraints of this economy are now:

$$\dot{K}(t) = i - \delta K \qquad [2.12]$$

$$\dot{R}(t) = \Omega - aY(K) - \gamma R \qquad [2.13]$$

where the change in the stock of man-made capital is defined as the difference between investment and depreciation (δ is the depreciation rate), and the change in the stock of environmental asset (e.g. forest biomass) is calculated as the difference between the level of improvements or maintenance Ω and the sum of degradation due to the economic activity $aY(K)$ and from natural causes γR.

The current-value Hamiltonian for this problem is:

$$H_c = U(C, A(R)) + \lambda_1[i - \delta K] + \lambda_2[\Omega - aY(K) - \gamma R] \qquad [2.14]$$

In order to bring this expression into what is actually measured by national accounting, [2.14] can be rewritten as:

$$H_c = S(C, A(R)) + C\frac{\partial U}{\partial C} + A(R)\frac{\partial U}{\partial A(R)} + \lambda_1[i - \delta K] + \lambda_2[\Omega - aY(K) - \gamma R] \qquad [2.15]$$

where [2.15] consists of consumer surplus of market goods and forest *amenities*, two terms that value the consumption of market goods and forest *amenities*, and two terms referred to the stock of man-made capital and environmental asset, respectively.

Total production of the economy is distributed between consumption, investment cost, $\varphi(i)$, and costs of maintenance, $\rho(\Omega)$, such that:

$$Y(K) = C + \phi(i) + \rho(\Omega) \quad [2.16]$$

The utility derived from the consumption of market goods can be calibrated such that $\frac{\partial U}{\partial C} = p$, being the monetary price of consumption. From the first-order conditions it is known that:

$$\lambda_1 = \frac{\partial U}{\partial C}\frac{\partial \phi(i)}{\partial i} = p\phi^{'}(i) = p_i \quad [2.17]$$

$$\lambda_2 = \frac{\partial U}{\partial C}\frac{\partial \rho(\Omega)}{\partial \Omega} = p\rho^{'}(\Omega) = p_\Omega \quad [2.18]$$

where p_i is the market price of investment and p_Ω is the market price of maintenance (defensive) expenditures.

Cairns proposes to calculate the linearized Hamiltonian[41], thus leaving out non-linearities (i.e. consumer surplus), and to evaluate consumption of market goods and forest *amenities* at the margin (see, Cairns (2000, 2002)). Including consumer surplus is correct as long as one also includes it for market goods and services. However, most theoretical approaches purge consumer surplus for market goods and services when interpreting NNP as a measure of social welfare. To quote Cairns (2001: 68), "the part of consumers’ surplus which is attributable to *amenities*, should not be part of green NNP, just as the part which is attributable to marketed goods is not a part of traditional NNP. In general, consumers’ surplus may not easily be divided among marketed and environmental goods" (see, Cairns (2008) and Caparrós (2010) for a similar argument).

Hence, [2.15] is now equivalent to NNP:

$$NNP = H_c = pC + A(R)\frac{\partial U}{\partial A(R)} + p_i[i - \delta K] + p_\Omega[\Omega - aY(K) - \gamma R] \quad [2.19]$$

[41] Linear measures permits aggregating results from small economic units at the shadow prices of the economy. The same is done in traditional NNP using market prices as proxies for shadow prices (Cairns, 2001). In this way NNP corresponds to a linear, and hence additive, index of intertemporal welfare (Caparrós et al., 2015).

The model by Cairns (2003) differs from the earlier in that aggregate social utility is a function of consumption, stock of environmental *amenities* or *disamenities* and labor exerted (a disutility). Moreover, the change in the stock of environmental assets is calculated as $\dot{R} = g(R) - h$, i.e. as the difference between growth and harvest.

On the other hand, the approaches taken by Vincent (1999a) and Matero and Saastamoinen (2007) share the same purpose and are very similar to one another. However, they are relatively different from Cairns' models as more forest-economy interactions are considered. Though apparently more comprehensive, the existence of so many interactions makes these models less flexible when it comes to adapting them to specific cases and different conditions. Furthermore, they make it all the more difficult to solve the necessary conditions of the problem. Still, an interesting contribution of these frameworks regarding the previous ones is the treatment of changes in the value of stocks, which opens the way to a discussion (we will come back to that later).

Vincent's (1999a) framework is quite complex, as there are five factors of production used by five different economic sectors to produce four goods and services. Production sectors are: forest management, F, logging, W, households' collection of non-timber products, N, land clearing (deforestation), D, and the rest of the economy, O. These sectors use labor, man-made capital, agricultural land, *I*, forestland, F [42], and emissions of air pollutants other than CO_2, e [43], to produce logs (wood, Z_W), non-timber products, Z_N, newly cleared land, Z_D, and to represent output in the rest of the economy. This output can be consumed, O_C, invested in man-made capital used both in the production of market goods and in forest management ($i_o + i_F$), or used in logging and forestry as intermediate input ($O_W + O_F$).

Households derive utility from consuming market goods, non-timber products and forest *amenities*; while they do not derive disutility from pollution. Concerning the change in stocks, the main difference with respect to Cairns (2001, 2003) is the inclusion of the following resource constraints:

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[42] The current area of agricultural land is equal to total land area minus the current forestland. Note also that F denotes both a sector and a variable, and the same is true for O.

[43] This model ignores transboundary pollution problems such as acid rain.

$$\dot{K}_F = i_F - \delta K_F \qquad [2.20]$$

$$\dot{I} = Z_D \qquad [2.21]$$

$$\dot{F} = -Z_D \qquad [2.22]$$

$$\dot{R} = g(L_F, K_F, O_F, R, F, e) - Z_W \qquad [2.23]$$

$$\dot{P} = \frac{dCO_2^{ROW}}{dt} - \alpha\{g(.) - Z_W\} \qquad [2.24]$$

where P is the stock of CO_2 in the global atmosphere. It is equivalent to average global CO_2 concentration. The change in the stock of land implies that forest area decreases, while agricultural area increases due to deforestation. The change in the timber stock equals the difference between growth and harvest, as in Cairns (2003), while defining the growth function differently. Finally, the change in the stock of atmospheric CO_2 is given by the difference between net emissions by the rest of the world and net sequestration in the country's forests.

On the basis of the Maler's (1991) approach, Vincent shows that the linearized Hamiltonian for the maximization problem provides an approximation to NDP, and that the latter should be adjusted in the following ways:

(i) Adjustments to the level of GDP and NDP

(ii) Adjustments to the sectoral composition of GDP

Concerning the former, adjusted NDP would read as follows:

Adjusted NDP =

Conventional GDP + *Non-market values to be added to GDP* −

Depreciation of man-made capital + *Net accumulation of natural capital*

Conventional GDP is $p_O(O_C + i_O + i_F)$, where p_O is the price of market goods. It should be adjusted by incorporating non-market goods and services, such as non-timber products and forest *amenities*, $p_N Z_N + U_F F$, where p_N is the price of non-timber products and U_F is the marginal utility of consumption of *amenities*[44].

Conventional NDP, which is equal to GDP minus depreciation of man-made capital $p_O(\delta K_0 + \delta K_F)$, should be adjusted to reflect changes in the value[45] of land (more specifically, land converted from forest to agriculture), timber stock and the stock of atmospheric CO_2. Net accumulation[46] is thus obtained by multiplying the variation in natural capital stocks in physical terms by the adjoint (costate) variable of these stocks, which can be interpreted as the marginal shadow price of capital stocks (this issue will be discussed in more detail below).

Concerning the second type of adjustments, Vincent (1999a) links conventional GDP to the sum of the value added (VA)[47] in industry, agriculture, logging and forest management:

Conventional GDP =

Conventional VA in logging + Conventional VA in tourism +

Conventional VA in industry + Conventional VA in forest management

In the first two cases, VA is expressed as the sum of payments to factors of production; whereas in the last two cases it is calculated as the difference between revenue from sales of output and expenditure on intermediate inputs. The question is how to reallocate VA for services provided by forests. For example, this model assumes that forests benefit other sectors but do not receive goods and services from the rest of sectors. Still, the level of GDP remains equal.

Matero and Saastamoinen (2007) draw up a similar framework on forest-economy interactions to that of Vincent (1999a), while adapting it to some specific features of Finish forests. For example, in addition to including emissions of air pollutants as a factor of production, they also consider emissions of pollutants to water, which will be increased as the production of logs is higher. The stock of nutrients pollutants in water

[44] Only *amenities* independent form markets, such as existence values of biodiversity, are considered.

[45] Note that net accumulation, and not simply reductions in value of forest-related assets, is included.

[46] Unexpected capital gains and losses are not included, as there are neither exogenous technological changes nor exogenous changes in prices.

[47] Without including taxes and subsidies, gross VA equals GDP.

negatively affects the production of market goods. Furthermore, non-timber products are considered as intermediate inputs in the production process.

While the aim of the Social Planner remains the same, in this model households explicitly derive disutility from the stock of threatened species representing the loss of existence values[48]. Resource constraints of the economy now refer to changes in the stock of: man-made capital (equations [2.12], [2.20]), land (equations [2.21], [2.22]), timber, threatened species, atmospheric CO_2 , soil nutrient and nutrients in water. Changes in timber stock are calculated as the difference between growth and the sum of harvest and natural mortality; while pollution stocks are defined as the difference between emissions from harvest and from decaying wood and the sum of net sequestration in the forest and decay in atmosphere. Changes in the stock of threatened species are given by the difference between the sum of cutting effect and decay and mortality of living trees. The soil nutrient stock is increased by the decay of dead wood and by deposition from air pollution and decreased by emissions to water from logging and forest management. Finally, the change in the stock of nutrients in water is defined as the difference between emissions from forests, deposition from air and internal load and sedimentation.

Adjusted NNP is now calculated as:

Adjusted NNP =

Conventional VA in logging + Conventional VA in tourism +
Conventional VA in industry + Conventional VA in forest management +

Non-timber services and (loss of) existence values –
Depreciation of man-made capital + *Net accumulation of natural capital*

Conventional GDP

Compared to Vincent (1999a), VA in agriculture has now been replaced by VA in tourism[49]. VA can also be expressed either as the sum of payments to factors of

[48] This argument is not, however, so far from the Vincent's (1999a) one as households derive utility from directly consuming forest *amenities*, which only refers to existence values associated with biodiversity.

[49] However, services provided by forest have been reallocated to other sectors in the same way.

production (VA in logging, tourism and industry) or as the difference between revenues from sales and expenditure on intermediate production (VA in forest management).

2.4.3.1. Empirical adjustments

As regard adjustments to conventional GDP, Vincent (1999a) points out that the value of non-timber products is proportional to the opportunity cost of labor used in collecting them, i.e. the result of multiplying the shadow price of labor by the amount of labor employed[50]. Matero and Saastamoinen (2007) also follow this line and use the average net hourly wage rate as a proxy for the value of time used in the production of (decorative) lichen, reindeer management and Christmas trees. By contrast, the existence value associated with forest *amenities*, such as biodiversity conservation, is derived by Vincent (1999a) through a contingent valuation survey, rather than through travel cost or hedonic pricing techniques. Using his own words, conservation is completely independent of marketed inputs and outputs. Compared with this, we offer an alternative solution (the SEV method) whereby markets are simulated to determine exchange values of non-market ecosystem services. Caparrós et al. (2003) and Campos and Caparrós (2006) also apply this method to pasture resources, game, forest recreation, carbon fixation and conservation in Spain. Matero and Saastamoinen also simulate the value of outdoor recreation activities, while not applying the SEV method.

As for adjustments to conventional NDP, net accumulation has been calculated by Vincent (1999a) and Matero and Saastamoinen (2007) as the physical variation in capital stocks times the corresponding adjoint variable (shadow price), the latter being expressed in terms of future returns to the natural capital stock (Caparrós et al., 2003). The treatment of timber stock is especially relevant for this dissertation. That is why we will dwell on this aspect further.

Vincent (1999b) shows that, for a forest where logging is done only once (at moment T), there are at least two correct methods for including net accumulation in the national accounts: (i) El Serafy variation, and (ii) net-price method.

The first method is so called, as it is a generalization of the method proposed by El Serafy for non-renewable resources. It differentiates between mature forests in which

[50] That said, it is important to mention that this is only possible under the condition that households have free access to the forest.

logging time is reached and immature forests in which this time has not yet come[51]. For mature forests, net accumulation tends toward current resource rent (with negative sign). In subsequent years, there will be a positive net accumulation, as the species is approaching the time of logging[52]. Thus, forestry supposes that the probability of harvesting is zero until the moment in which the tree reaches the diameter-age associated to age *T*. At that moment, the probability of harvesting is one. This would imply knowing future forestry for each of the species available[53].

The second method assumes that logging follows the Faustmann-Hartman 'optimal rotation'. It maximizes the net present value behind an infinitive chain of planting cycles, rather than a unique planting cycle[54]. This means that there is an opportunity cost of land rents and that the method involves marginal rents (see, Olschewski and Benítez (2005)). It does not, however, produce precise results if all trees in a given stand are the same age-class[55].

As appeared from Vincent (1999a), the adjoint variable for timber stock is a proxy for the marginal stumpage value (marginal 'net price'). This is because 'net-depletion' method does not take into account the age-class of the forest, thus not affecting the price of timber. It therefore obviates the time lags that occur between timber harvests. The fact that the age-class does not affect the price is not what normally happens, so

[51] When this differentiation is made, 'net-depletion' method overstates both the decrease in forest value that occurs when mature forests are logged and the appreciation of immature forests due to growth (Vincent, 1999b).

[52] The methodology set out by Caparrós et al. (2003) confirms the results of El Serafy variation method, which is a particular case when all trees are cut down upon reaching age *T*.

[53] This probability may be estimated building on historical wood removals, thus determining the conditional probability that a living tree of the diametric-age *d* is harvested in the diametric-age *j*, where $j > d$ (Caparrós et al., 2003).

[54] As pointed out Benítez-Ponce (2005: 20), "it might happen that for some standing forests the spillover amenity benefits might be so great that it would not be economically feasible to harvest the forest at any time in the future. This would be represented by an infinite Faustman-Hartman rotation length".

[55] Cairns (2001) shows that if the forest is an even-aged stand on a given area, the cut rotation period is optimal and there is also a single species, it can easily be a stand with multiple ages with selective cutting of tree of age *T* on a sustained basis. Consequently, the value obtained through this method is relatively similar to that obtained discounting the future.

Caparrós et al. (2003) propose to project values for different age-classes into the future by discounting future returns (we will come back to that later).

According to the criteria of conventional national accounts explained above, Caparrós et al. (2003) calculate the stock values in the capital balances (i.e. at the beginning and at the end of the accounting period) discounting future returns to the corresponding capital stocks. In this way, net accumulation of man-made and natural capital is obtained by multiplying the physical variations by the shadow price (discounted future returns). Another difference compared to the Vincent's (1999a) model is that price of timber is highly dependent on the age-class of the forest. Hence, the growth in forest not only produces more tons of timber, but also a revaluation of the stock existing at the beginning and at the end of the accounting period due to the changes in the 'quality' of the timber when passing from one diameter class to the next. This change in the value must be incorporated into the asset accounts.

2.4.4. Pollution

There is a perception that national accounting overestimates GNNP, as it does not account for the damage to the environment from pollution emissions. Attempts have been made to include pollution damages and abatement costs (or defensive expenditures) in the national accounting literature. The basic idea is that the value of environmental damage should be deducted from green GDP in order to arrive at GNNP, and that defensive expenditures must be treated as intermediate consumption, rather than as final demand (Hamilton, 1995).

Hartwick (1990) explicitly extends Weitzman's approach to environmental damages caused by pollution. The stock of pollution is an input into the production function and affects it negatively, as the higher the level of pollution, the lower production will be. But at the same time, more production generates higher levels of pollution, so it is necessary to determine the net pollution increments, i.e. the difference between pollution due to production and the amount of pollution that evaporates by natural environmental stock regeneration. This means that pollution is controlled indirectly by output decisions. We also consider this assumption in the model developed in the third chapter. However, changes in the pollution stock may also appear in the utility function (Hartwick, 1993). In both cases, abatement costs must be deducted from GNP.

Mäler (1991) constructs a similar model where the quantity of residuals generated is an argument within the production function and where firms can buy pollution control services offered by pollution control firms. Households can also improve their environment by defensive expenditures. These expenditures have often been thought of as a proxy of environmental damage. This means that one may subtract defensive expenditures, rather than environmental damage. However, as pointed out by Mäler, this is true only if these expenditures are perfect substitutes to environmental services.

In a related field, while not part of green national accounting, Feng et al. (2002) examine land use changes driven by a climate change policy. The optimal patterns of carbon sequestration, including carbon emissions and sequestration simultaneously, are analyzed for a scenario of instantaneous growth of new reforestation. This is particularly restrictive for the forestry sector, as many species need long periods of time to reach maturity. Caparrós (2009) relaxes this assumption and considers that when land is converted, carbon sequestration takes place at two moments in time. Caparrós and Zilberman (2010) include the intensive margin of land use climate policies, represented by the choice of species each of them growing differently. Thus, the optimal control problem allows Social Planner to choose the type of species to plant and to take into account, in each moment in time, the growth performed by all trees in the past.

Assuming the basic premises of theoretical studies on forests above-mentioned makes it possible to estimate GNNP using information of the current year. However, although this scenario has been widely adopted, it is unrealistic. Hence, building on the model developed by Caparrós and Zilberman (2010) in the context of carbon sequestration, a model incorporating the growth of different vintages in the forests using Volterra integral equations is developed in the third chapter of this dissertation. Each portion of forest planted at any period s ($s<t$) continues to grow at a rate that depends on the type of species chosen at time s and on the age of the forest, i.e. the time lag between s and t. This involves having future values.

2.4.5. Beyond the Weitzman's approach: limitations and alternative proposals

In the theoretical model set up in the following chapter, we follow the tradition initiated by Weitzman (1976) and subsequently extensions. Even so, we are aware that there is another stand of literature (Hamilton and Clemens, 1999; Dasgupta and Mäler, 2000; Arrow et al., 2003a,b; Dasgupta, 2009; Arrow et al., 2012; Fenichel and Abbott, 2014) that raises questions about the usefulness of Weitzman's approach and its derivatives. Interesting though these proposals may be, we have, nevertheless, decided to follow the approach pioneered by Weitzman (1976), and therefore to set aside these criticisms, as it is closely related to NNP and because most of the results have been derived within this tradition[56] (see, for example, previous models on forests). In any case, we will explain what this alternative proposal is about. The main criticisms concern the fact that (i) utility function is linear, (ii) economy is on the optimal path, and (iii) there is no exogenous technological change.

With regard to the first assumption, the argument goes that utility function should be concave in consumption (see, Kemp and Long (1972)). This is widely accepted, as assuming concavity is more convenient for applied works (Vincent, 2000). So, why are linear utility functions so common in economics? Welfare measures must be transformed into real terms if a static welfare equivalent is being sought. The conventional way to do so is to linearize the current-value Hamiltonian[57], which is the same as normalizing the price of consumer goods to one (see previous models by Cairns (2001, 2003)). Thus, utility and money metrics are equivalent (Aronsson et al., 2004: 34). Conversely, if the utility function is concave, the current-value Hamiltonian and the linearized current-value Hamiltonian are not the same and the welfare interpretation of the latter remains unclear. Weitzman (2000) attaches the problem by calibrating the instantaneous utility function so that the normalized function was proportional to the money-metric value of consumption[58]. That is, if linearizing the Hamiltonian is interpreted as money metricizing the instantaneous utility function, the Hamiltonian is not measured in utility terms, but with aggregated consumption as numeraire. He also made a generalization of this approach by allowing the possibility to have multiple

[56] Yet, the key results that we obtain in the model could also be derived within the framework pioneered by Dasgupta and his co-authors.

[57] However, once linearity is abandoned, we have to work with consumer surplus (Weitzman, 2001).

[58] Any positive affine transformation of the utility function induces the same welfare, i.e. the instantaneous utility function $W(C) = aU(C) + b$ gives a welfare ordering that does not depend on $a > 0$ and b (Weitzman, 2000: 59), which are constant (Asheim, 2004: 37-42).

consumption goods[59], including environmental goods and services, as they also contribute to the improvement of the 'standard of living'.

Nevertheless, one needs to enable intertemporal index comparisons. This led Weitzman (2001) to derive an exact welfare measure in real terms. As the utility function is quasi-linear (concave) in income, the change in the present discounted future utility is defined as the change in real income plus consumer surplus, which is also a quasi-linear utility function (Weitzman, 2000) representing the information lost by linearizing the Hamiltonian (Aronsson et al., 2004). Welfare function therefore consists of monetary and utility terms, making it necessary to take out the marginal utility of income. To that end, Weitzman (2001) introduces an 'ideal' market-basket price index, and develops a 'unified theory' according to which consumer surplus theory and index-number theory are related. Imposing the money-metric normalization is equivalent to using Marshallian consumer surplus, which is the appropriate measure of social welfare when income effects do not exist.

Concerning the second assumption, the resource allocation mechanism of an economy is considered to be optimal by the Weitzman's approach, thereby implementing an efficient path that maximizes welfare (Asheim, 2003). Although it is very popular among economists, it is at the same time unrealistic in the current context where property rights and other institutional failures exist. This is particularly relevant when it comes to natural resources and environmental management regimes (Vincent, 2000). Dasgupta and his co-authors contend that economies do not pursue optimal policies, so NNP cannot be interpreted as a measure of social welfare. Since the resource allocation is sub-optimal, the evaluator must carry out an economic program in order to enhance the path of consumption, production and capital accumulation. Hence, welfare will increase over time if a small investment project[60] improves NNP measured in local shadow prices (Kriström, 1996; Kriström and Skånberg, 2001; Aronsson et al., 2004: 43). In an optimal and competitive economy where the set of taxes and subsidies

[59] As explained above, practitioners use the aggregate market value of consumption as the standard indicator when estimating current consumption, and thus social welfare, in the national accounts.

[60] Focusing on small projects is not trivial considering that "if the alterations in economic activities were not small (i.e. if they were to affect the accounting prices), the appropriate index of social well-being would be non-linear. This is because the index would then have to include changes in consumers' and producers' surpluses, and changes in income distributional weights" (Dasgupta et al., 1997: 129).

supports a full optimum, market prices are equivalent to local shadow prices. However, in imperfect economies, market prices are solely a good approximation for shadow prices for some goods and services (Dasgupta, 2009). This means that it would be enough to empirically quantify direct consumption values of the environment and values associated with the stock of natural capital (Vincent, 2000).

This is related, but not identical, to another argument that opposes Weitzman's proposal: NNP should no longer be used to make comparisons across time and space and to evaluate long-term policies, but to assess short-term policies, even if the problem is non-convex. Thus, a short-term policy reform increases social welfare if and only if NNP measured in local accounting prices increases (Dasgupta and Mäler, 2000). Even so, linear indexes in quantities must be developed when it comes to making intertemporal and cross-country comparisons, as well as when checking if policy reforms result in sustainable development[61.] Weitzman maintains that NNP, measured at shadow (accounting) prices, is such index in the context of a closed economy[62]. Even if the utility function is concave, changes in NNP still reflect changes in welfare (Asheim and Weitzman, 2001). Critics propose to work with economic forecasts and to focus on comprehensive wealth instead. It is a state variable, rather than a flow variable, which takes into account total stock of composite goods and natural resources valued at shadow prices. In either case, it is not the same as social welfare, but both concepts respond in the same direction to perturbations (Dasgupta, 2009). In line with comprehensive wealth, comprehensive investment, or equivalently genuine saving[63], has emerged as a widely used indicator of sustainable development (Oxley et al., 2014)

[61] If these indexes were not used, future changes in consumer surpluses would have to be estimated.

[62] The choice of metrics does not have any impact on welfare measurement, so NNP growth entails welfare improvements as a result of endogenous changes in shadow prices (Li and Löfgren, 2002). However, if consumption price changes, this relationship is not so clear-cut. There is a case where NNP growth indicates a welfare improvement; hence it requires the Divisia consumption price index (Sefton and Weale, 2006; Asheim and Wei, 2009). Then, growth in real NNP can be used for intertemporal welfare comparisons (Asheim and Weitzman, 2001), thus differing from the claim made by Dasgupta and Mäler (2000). All this, however, entails having positive interest rate (Li and Löfgren, 2006).

[63] The variation in comprehensive wealth is measured by genuine savings (Pearce and Atkinson, 1993, 1998; Hamilton and Clemens, 1999; Hamilton and Ruta, 2009). It is simply net savings (gross savings less depreciation on man-made and environmental capital), measured at shadow prices.

as it measures the rate at which intergenerational social welfare changes over time (i.e. the present discounted value of changes in consumption) (Dasgupta, 2009)[64].

The last assumption regarding stationary technology is one of the most restrictive assumptions with profound and far-reaching implications. If technological change is not attributable to any specific production factor, a non-time-autonomous model emerges and the welfare significance of NNP becomes more complex, as it is difficult to predict the variation of utility[65]. NNP equals stationary equivalent of future consumption possibilities without the 'Solow residual' (i.e. the non-attributable technological change), but the inclusion of such residual would involve a larger NNP. Then the current-value Hamiltonian should also incorporate the present value of marginal technological change in order to be a measure of social welfare (Aronsson and Löfgren, 1998b). NNP, as currently defined in [2.4], only takes into account endogenous changes to capital stock. Exogenous technological progress may take different forms, including exogenous changes in prices, so real NNP should be modified to include changes in the values of capital stocks, namely anticipated capital gains. They should also be taken into account if interest rate (or utility discount rate) is volatile and technology does not exhibit constant returns to scale[66] (see, Wei (2012)).

$$NNP(t) = C(t) + p(t)\frac{dK}{dt}(t) + \frac{dp}{dt}(t)K(t) \qquad [2.25]$$

Under these assumptions, the conclusion to be drawn is that wealth-equivalent income underestimates welfare-equivalent income (Asheim, 1997) and sustainable income. In the latter case, as prices are not constant, there is a loss of present values when turning the current consumption path into a constant consumption path.

In the third chapter of this dissertation, we extend our benchmark framework from which it follows that the growth of natural capital (forest biomass) is no longer constant.

[64] Achieving an overall sustainable development at time t requires at least one of these three conditions: (i) comprehensive wealth should be non-declining at t, (ii) comprehensive investment should be non-negative (Arrow et al., 2003b, 2012), or (iii) the value of net changes in the flow of consumption plus the change in comprehensive investment should be non-negative.

[65] In this case, the current value Hamiltonian will not represent a static equivalent of welfare along the optimal path (Aronsson et al., 1997: 54).

[66] Note that NNP only coincides with wealth-equivalent income if production technology exhibits constant return to scale (Vincent, 2000)

This means having unanticipated capital gains, as the value of forest biomass at moment t will differ from the value of forest biomass at moment s (for $s \leq t$), so at each period of time, the value will change and depend on the species chosen and on the age of such species. This interpretation is similar to that described above for technology progress.

CHAPTER 3. GREEN ACCOUNTING AND TERRESTRIAL ECOSYSTEMS: FOREST CAPITAL AND NON-MARKET GOODS AND SERVICES

3. GREEN NATIONAL ACCOUNTING AND TERRESTRIAL ECOSYSTEMS: FOREST CAPITAL AND NON-MARKET GOODS AND SERVICES[67]

This chapter presents an intertemporal optimization model that extends Weitzman's results to the forestry sector[68]. More specifically, this theoretical approach is similar to those of Cairns (2001, 2003), while nevertheless showing that one of the results is not applicable to this sector. We prove that GNNP can continue to be measured using only values of the current year when there is a single species that grows following an exponential function with constant decay and there is no felling. By contrast, if different types of species exist, GNNP cannot be estimated using values of the current year, as in Vincent (1999a) and Cairns (2003), as there are future values.

We also discuss the need to expand GNNP to incorporate non-market services such as public forest recreation or biodiversity conservation. Our results confirm that consumer surplus (or any Hicksian variation) must not be used for valuing non-market goods and services, once they have been taken out for market goods and services. Instead, these services should be incorporated into national accounting using exchange values. In doing so, we provide theoretical support for the SEV method explained in the previous chapter. We first characterize exchange values for the case of perfect competition, although we then extend the analysis to cover monopoly and monopolistic competition. The latter are especially relevant for recreational use of National Parks which are unique, or at least rare.

[67] This chapter is based on Caparrós and Ruiz-Gauna (2016).

[68] As explained in the previous chapter, there is a critical literature on the Weitzman's approach that argues that the adequate linear index of welfare is not NNP, but 'comprehensive wealth'. As the goal of this chapter is to expand NNP by including ecosystem services, we do not follow this approach.

3.1. Benchmark model

This closed economy consists of two sectors: the forestry sector and the rest of the economy. As described in previous theoretical models, $K(t)$ is the stock of man-made capital, $R(t)$ is the stock of natural capital (forest biomass, for example), and $P(t)$ is the stock of atmospheric pollution[69]. In order to maintain the problem as simple as possible, we assume that there is no man-made capital in the forestry sector. Investment in man-made capital is represented by $i(t)$ and the extraction of forest biomass is given by $h(t)$. Each unit of production is emitting *e* units of carbon (which is considered to be the sole pollutant).

The Social Planner's problem is:

$$\max_{i(t),h(t)} \int_0^\infty U\left(C(t), A(R(t))\right) e^{-rt} dt \qquad [3.1]$$

subject to:

$$\dot{K}(t) = i(t) - \delta K(t) \qquad [3.2]$$

$$\dot{R}(t) = g(R(t)) - h(t) \qquad [3.3]$$

$$\dot{P}(t) = e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t) \qquad [3.4]$$

$$X(0) = X_0, \lim_{t\to\infty} X(t) \geq 0 \text{ for } X = K, R, P \qquad [3.5]$$

where $A(R(t))$ is a function that collects non-market benefits (*amenities*) provided by forests, which depend on forest biomass, as in Cairns (2001). Furthermore, and in order to simplify the analysis, we assume that these *amenities* do not influence the production function[70]. Equation [3.2] indicates that the change in the stock of man-made capital equals the difference between gross investment and depreciation. Equation [3.3] indicates that the change in the stock of forest biomass is determined by the difference between trees' growth, which is given by the function $g(R(t))$ and the selective

[69] Unlike in the second chapter, here we need a source of time-dependence, as the derivation of the problem will not only include values at one moment in time *t*, as we will see below.

[70] This way of dealing with *amenities* is common for the existence value of biodiversity, but we extend it to other non-market ecosystem services such as public forest recreation.

extraction of wood (measured in tons). Equation [3.4] shows that the change in the stock of pollution is given by the difference between emissions and the sum of the amount of carbon sequestered in forests and decay in atmosphere (σ denotes the decay rate). Equation [3.5] represents the appropriate transversality conditions.

The economy's total production at time *t*, $Y(K(t), h(t), P(t))$, is a function of the stock of man-made capital, forest biomass extracted and the stock of pollution. As we are in a general equilibrium model, final production is distributed between consumption of market goods and services, the investment cost, $\phi(i(t))$, and the cost of extracting forest biomass, $\psi(h(t))$, such that:

$$Y(K(t), h(t), P(t)) = C(t) + \phi(i(t)) + \psi(h(t)) \qquad [3.6]$$

The current-value Hamiltonian[71] for this problem is:

$$\begin{aligned} H_c(t) = {} & U[C(t), A(R(t))] + \lambda_1(t)[i(t) - \delta K(t)] + \lambda_2(t)[g(R(t)) - h(t)] \\ & + \lambda_3(t)[e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t)] \end{aligned} \qquad [3.7]$$

In order to bring this expression into what is actually measured by national accounting, [3.7] can be rewritten using the same affine transformation as Cairns (2001, 2002):

$$U[C(t), A(R(t))] = S[C(t), A(R(t))] + \frac{\partial U(t)}{\partial C(t)} C(t) + \frac{\partial U(t)}{\partial A(R(t))} A(R(t)) \qquad [3.8]$$

so that [3.7] becomes:

$$\begin{aligned} H_c(t) = {} & S[C(t), A(R(t))] + \frac{\partial U(t)}{\partial C(t)} C(t) + \frac{\partial U(t)}{\partial A(R(t))} A(R(t)) + \lambda_1(t)[i(t) - \delta K(t)] \\ & + \lambda_2(t)[g(R(t)) - h(t)] + \lambda_3(t)[e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t)] \end{aligned} \qquad [3.9]$$

[71] Rather than using the Hamiltonian, we work with the current-value Hamiltonian, as in Cairns (2001, 2003), as the presence of the discount factor only adds complexity to the derivatives of the maximization problem. In any case, it can be proven that if we had used the Hamiltonian, the solution would not have been modified, as the current-value Hamiltonian is defined as a Hamiltonian that is free of the discount factor (Chiang, 1999: 210).

where [3.9] has now three clearly differentiated parts: i) $S[C(t), A(R(t))]$, which represents consumer surplus of market and non-market goods and services, ii) two terms that value consumption and non-market benefits by using exchange values (as we will see below), and iii) the last three terms which refer to the stock of man-made capital, natural capital (forest biomass in our case) and pollution, respectively.

It is possible to continue working with the utility as numeraire, but the conventional 'trick' consists in calibrating the utility derived from the consumption of market goods and services such that $\frac{\partial U(t)}{\partial C(t)} = p(t)$, $p(t)$ being the monetary price of consumption, which is considered to be constant.

From the necessary conditions (see Appendix A), we also know that:

$$\lambda_1(t) = p_i(t) \qquad [3.10]$$

$$\lambda_2(t) = B_1(t)\eta + p_w(t) - p_h(t) \qquad [3.11]$$

$$\lambda_3(t) = B_1(t) \qquad [3.12]$$

with

$$p_i(t) = \frac{\partial \phi(i(t))}{\partial i(t)} p(t) \qquad [3.13]$$

$$p_w(t) = \frac{\partial Y(K(t), h(t), P(t))}{\partial h(t)} p(t) \qquad [3.14]$$

$$p_h(t) = \frac{\partial \psi(h(t))}{\partial h(t)} p(t) \qquad [3.15]$$

$$B_1(t) = \left[\frac{(r+\delta)\frac{\partial \phi(i(t))}{\partial i(t)} - \frac{\partial Y(K(t), h(t), P(t))}{\partial K(t)}}{\frac{\partial e(K(t))}{\partial K(t)}} \right] p(t) \qquad [3.16]$$

where $p_i(t)$ is the market price of investments in man-made capital, $p_w(t)$ is the market price of the forest biomass (wood) as a factor of production, and $p_h(t)$ is the market price (cost) of extracting wood. $B_1(t)$ relates real profits (or losses) for each unit of capital[72] to the responsiveness of emissions to a change in an additional unit of capital[73].

These conditions allow us to rewrite [3.9] (see Appendix A) as:

$$H_c(t) = S[C(t), A(R(t))] + p(t)C(t) + \frac{\partial U(t)}{\partial A(R(t))} A(R(t)) + p_i(t)[i(t) - \delta K(t)]$$
$$+ (p_w(t) - p_h(t))[g(R(t)) - h(t)] + B_1(t)[e(K(t)) - \sigma P(t)] \qquad [3.17]$$

The last 'trick' consists in 'linearizing' [3.17] (Cairns, 2001, 2003, 2008), which means taking out consumer surplus, as it does not depend linearly on $C(t)$ and $A(R(t))$ (see, however, Dasgupta's (2009) criticisms to this procedure). Remember that, as explained above, we are removing consumer surplus for both market and non-market goods and services. Given that NNP is calculated by multiplying market prices by quantities of market goods and services, we define GNNP as:

$$\hat{H}_c(t) = GNNP(t) = p(t)C(t) + \frac{\partial U(t)}{\partial A(R(t))} A(R(t)) + p_i(t)[i(t) - \delta K(t)]$$
$$+ (p_w(t) - p_h(t))[g(R(t)) - h(t)] + B_1(t)[e(K(t)) - \sigma P(t)] \qquad [3.18]$$

The first term of [3.18] refers to the consumption of market goods and services at their market price, as entered in conventional national accounting. The second term refers to the consumption of non-market goods and services multiplied by their non-market value, which is given by its contribution to the utility of individuals (section 3.3. is entirely devoted to the valuation of this term). The third term refers to the

[72] The L.H.S. of the numertator is the real cost of acquiring one addtional unit of capital. When a unit of capital is acquired, the money that it would be obtained by investing costs of acquisition is being left. Moreover, capital depreciates. Thus, real cost would be determined by $(r+\delta)p_i(t)$. The R.H.S. is the value of marginal product of capital, also known as the marginal revenue product. It is defined as the change in total revenue that results from employing one more unit of capital, i.e, the market value of one additional unit of output. For a perfectly competitive economy, marginal revenue product diminishes as the quantity of capital employed increases because the marginal product of capital diminishes.

[73] As expalined in the second chapter, pollution is controlled by output decisions, as in Hartwick (1990).

depreciation of man-made capital (valued at its market price). The last two terms refer to net accumulation of forest biomass and pollution, respectively.

The term $(p_w(t) - p_h(t))[g(R(t)) - h(t)]$ in [3.18] is deceptively simple, as will be explained in the section below. Under the expression of GNNP which we have just derived, we would simply have to assess net growth (natural gross growth in the year minus extractions in the year) by its stumpage price (the value of wood as a factor of production minus extraction costs). This simplicity is due to the assumptions made (see, Repetto et al. (1989); Hartwick (1990); Hultkrantz (1992); Vincent and Hartwick (1997); Vincent (1999a); Kriström and Skånberg (2001); Caparrós (2010)).

3.2. Forest capital

We concluded the previous section by indicating that the fourth term in [3.18] was overly simple. If we could effectively use this term to calculate GNNP, we would have the obvious advantage of being able to estimate this indicator using only values of the current year. In this section, we relax the assumptions that yield this result and discuss their implication for green accounting (building on the model developed by Caparrós and Zilberman (2010) in the context of carbon sequestration).

We show that if the growth of a single species can be approximated by an exponential function with constant decay and there is no felling, GNNP can continue to be estimated by using only values of the current year[74]. Without a doubt, this scenario is restrictive. That is why we also analyze the implications of allowing different types of species to exist. This assumption means that there are future values so that it is not possible to estimate GNNP by using values of the current year[75].

[74] If we would extend this analysis by assuming that after *T* years all trees are felled, we would no longer deal with a conventional optimal control model, but rather with a modified model in which there would be variables at time *t* and time-delayed variables. In any case, GNNP only depends on two points in time (we will come back to that later).

[75] While we are aware that tree volume tends to grow following a logistic function (or other functions such as the Chapman-Richards, among others), we utilize exponential functions. Optimal control theory – starting with the classical Ramsey-Cass-Koopmans model – implicitly uses exponential functions due to the ease of which state equations can be derived. In fact, if we worked with logistic functions, state equations would take a different form than that adopted in optimal control models.

3.2.1. A single species and a constant decay exponential growth function

Let us first suppose that, instead of felling trees, the forest grows without human intervention following a constant decay exponential function. The amount of growth (trees are treated as 'growth' machines) produced at time *s* is called *x(s)*, which continues to generate growth over time at a decaying rate. The cost of planting is $\varsigma(x(s))$ and the growth function of one tree planted at time *s* in any moment in time *t* (note that trees are 'planted' in every instant) is denoted *m*, according to the growth function detailed in the next definition:

Definition 1 ***Constant decay exponential growth function****: Assume that the growth function of one tree takes the following exponential form*

$$m(t,b,s)=G_0 e^{-b(t-s)}, \forall t \geq s \qquad [3.19]$$

where b and G_0 *are parameters. Hence, the growth of all vintages at time t, denoted by G(t)*[76]*, is given by:*

$$G(t)=\int_0^t x(s)e^{-b(t-s)}ds \qquad [3.20]$$

and the function that accumulates all the past growth – i.e. forest biomass R(t) – is:

$$R(t)=\int_0^t G(z)dz=\int_0^t\left(\int_0^z x(s)e^{-b(z-s)}ds\right)dz \qquad [3.21]$$

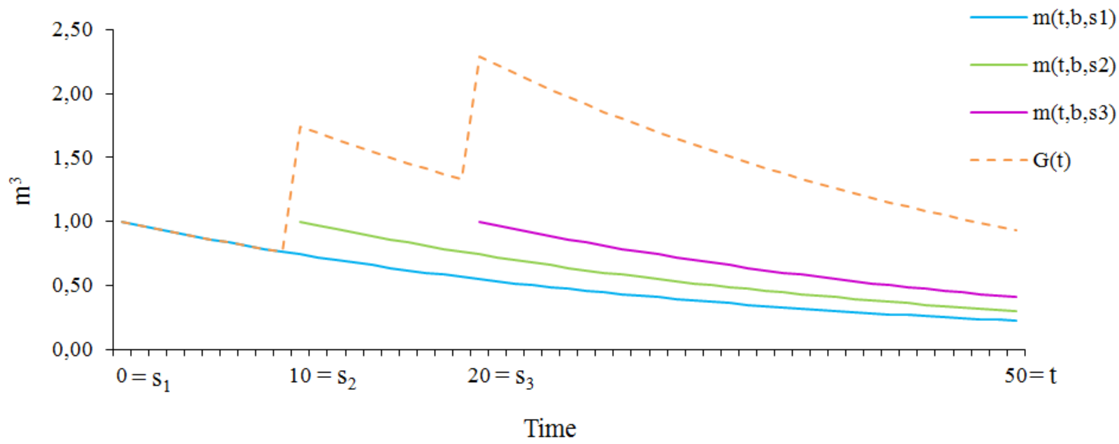
To simplify the analysis, we assume that $G_0 = 1$, as this has no impact on our results.

Let us see the argument graphically with a particular numerical example[77]. A reforestation process is implemented at three different points in time – $s_1 = 0$, $s_2 = 10$ and $s_3 = 20$ –, as shown in Figure 3.1. These new-planted trees (remember that trees belong to one and the same species) grow following the constant decay exponential function [3.19]. However, the growth of all vintages, $G(t)$, has a different shape. From $s_1 = 0$ to $s_2 = 10$, there is just one vintage, so $G(t)$ coincides with the growth of trees belonging to this vintage. This is no longer the case at $s_2 = 10$, as a new reforestation

[76] In the benchmark model, trees' growth is denoted by *g(t)*; while here it is denoted by *G(t)*.

[77] Values given to the function are: $m(t,b,s_1) = 1$, $s_1 = 0$, $s_2 = 10$, $s_3 = 20$, $b = 0.03$.

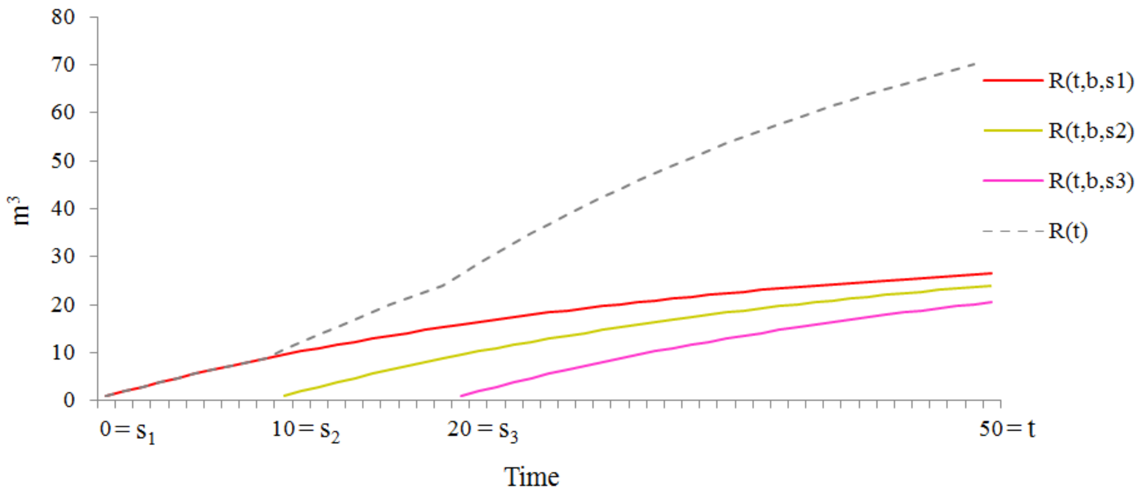
takes place. This means that two vintages are underway. Thus, the value of $G(t)$ is larger than before. Forest is again reforested at s_3 = *20* with the ensuring increase in the value of $G(t)$. Hence there are three vintages at this time. Because no more reforestations are undertaken, the value of $G(t)$ continues to decay from s_3 = *20* on.



Source: own elaboration

Figure 3.1. Growth of all vintages at time *t* when there is a single species and a constant decay exponential growth function

The reverse process occurs when it comes to the accumulated past growth (Figure 3.2). The growth of trees planted in the three vintages accumulates over time, so the value of $R(t)$ is continually growing. Moreover, at s_2 = *10* and s_3 = *20*, this value increases more than before as new trees are planted.



Source: own elaboration

Figure 3.2. Accumulated past growth at time *t* when there is a single species and a constant decay exponential growth function

The Social Planner's problem is now:

$$\max_{i(t),x(t)} \int_0^\infty U(C(t), A(R(t)))e^{-rt}dt \qquad [3.22]$$

subject to [3.2], [3.20], [3.21], the appropriate transversality conditions, and a modified version of [3.4], which is now defined as:

$$\dot{P}(t) = e(K(t)) - \eta \dot{R}(t) - \sigma P(t) \qquad [3.23]$$

The economy's total production at time t is given by[78]:

$$Y(K(t), x(t), P(t)) = C(t) + \phi(i(t)) + \varsigma(x(t)) \qquad [3.24]$$

In order to transform this problem into a standard optimal control model, we can take the time derivatives of [3.20] and [3.21] such that:

$$\dot{G}(t) = x(t) - b\int_0^t x(s)e^{bs-bt}ds = x(t) - b\int_0^t x(s)e^{-b(t-s)}ds = x(t) - G(t) \qquad [3.25]$$

and

$$\dot{R}(t) = \int_0^t x(s)e^{bs-bt}ds = \int_0^t x(s)e^{-b(t-s)}ds = G(t) \qquad [3.26]$$

so we can now rewrite the Social Planner's problem as before, while replacing [3.20] by [3.25], [3.21] by [3.26] and [3.23] by

$$\dot{P}(t) = e(K(t)) - \eta G(t) - \sigma P(t) \qquad [3.27]$$

Thus, the current-value Hamiltonian of this problem is:

$$\begin{aligned} H_c(t) &= U[C(t), A(R(t))] + \lambda_1(t)[i(t) - \delta K(t)] + \lambda_2(t)G(t) \\ &+ \lambda_3(t)[e(K(t)) - \eta G(t) - \sigma P(t)] + \lambda_4(t)[x(t) - bG(t)] \end{aligned} \qquad [3.28]$$

From the necessary conditions (see Appendix B), we know that, apart from [3.10]:

[78] In order to simplify the analysis, let us assume that, in the two cases analyzed in this section, there is no cost of extracting forest biomass.

$$\lambda_2(t) = B_2(t)\eta + \left(p_{F_x}(t) - p_x(t)\right)[b + r] \quad [3.29]$$

$$\lambda_3(t) = B_2(t) \quad [3.30]$$

$$\lambda_4(t) = p_{F_x}(t) - p_x(t) \quad [3.31]$$

with

$$p_{Fx}(t) = \frac{\partial Y(K(t), x(t), P(t))}{\partial x(t)} p(t) \quad [3.32]$$

$$p_x(t) = \frac{\partial \varsigma(x(t))}{\partial x(t)} p(t) \quad [3.33]$$

$$B_2(t) = \left[\frac{(r+\delta)\frac{\partial \phi(i(t))}{\partial i(t)} - \frac{\partial Y(K(t), x(t), P(t))}{\partial K(t)}}{\frac{\partial e(K(t))}{\partial K(t)}} \right] p(t) \quad [3.34]$$

where $p_{Fx}(t)$ is the marginal revenue product of the amount of growth produced at time *t*. $p_x(t)$ is the market price (cost) of planting trees.

After affine transformation and purging consumer surplus, the 'linearized' current-value Hamiltonian and GNNP (see Appendix B) are:

$$\begin{aligned} \hat{H}_c(t) = GNNP(t) = p(t)C(t) + \frac{\partial U(t)}{\partial A(R(t))} A(R(t)) + p_i(t)[i(t) - \delta K(t)] \\ + \left(p_{F_x}(t) - p_x(t)\right)[rG(t) + x(t)] + B_2(t)[e(K(t)) - \sigma P(t)] \end{aligned} \quad [3.35]$$

In short, we show that if:

(I) there is a single species that grows instantaneously (as in the benchmark model), or

(II) there is a single species that grows following an exponential function with constant decay [3.19],

then we are back to an optimal control model where both the 'linearized' current-value Hamiltonian and GNNP depend only on values of the current year.

3.2.2. Different types of species

Building on Caparrós (2010) and Caparrós and Zilberman (2010), we now consider that each tree grows following a different exponential function. To simplify the analysis, we assume that there is a continuum of forest species[79] and that each species has a different $b(s)$, defined at the moment of planting. Only one type of species is planted in each instant. Still, we suppose that for all species $G_0(s) = G_0 = 1 \;\; \forall s$. Hence, different species only differ in their $b(s)$.

We also assume that trees are not felled, and are thus only planted to increase *amenities*. The result would also hold if trees were felled (just adding unnecessary complexity).

Let us now consider the case of a non-constant decay exponential function, as the one defined below:

Definition 2 ***Non-constant decay exponential growth function****: Assume that, at each moment of time s, the growth function takes the exponential form*

$$m(t, b(s), s) = G_0(s)e^{-b(s)(t-s)} \forall t \geq s \qquad [3.36]$$

and hence that the growth of all vintages at time t, G(t), is now given by:

$$G(t) = \int_0^t x(s)e^{-b(s)(t-s)}ds \qquad [3.37]$$

and that the function that accumulates all the past growth, R(t), is:

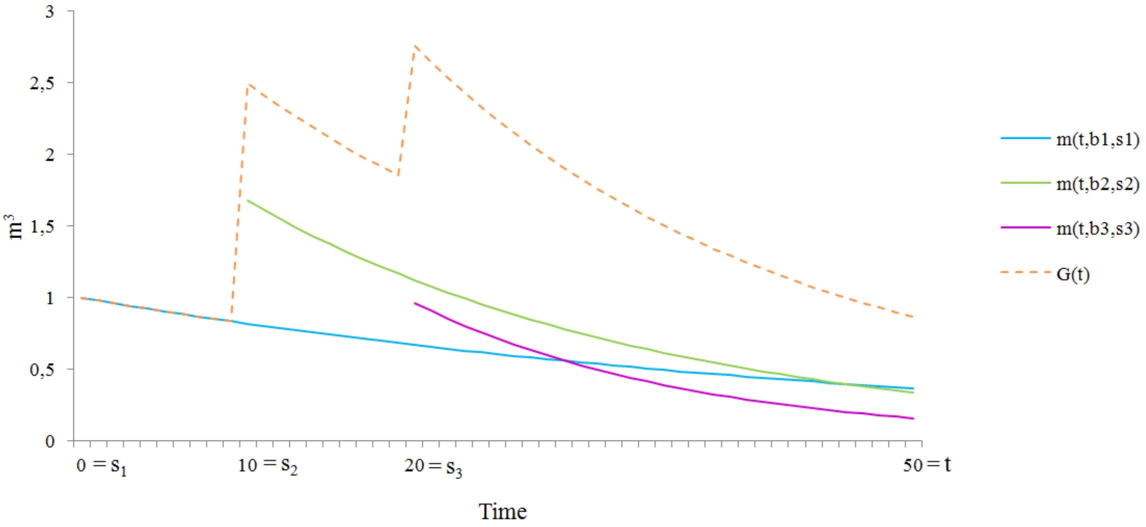
$$R(t) = \int_0^t \left(\int_0^z x(s)e^{-b(s)(z-s)}ds \right) dz \qquad [3.38]$$

Graphically[80], growth is no longer constant and differs among species (see, for example, the different growth paths between trees planted in the first and third vintages

[79] Although this is unrealistic, it allows us to stress our main point in a relatively straightforward model.

[80] In the figure, values given to the function are: $m(t,b_1,s_0) = 1$, $m(t,b_2,s_0) = 2.5$, $m(t,b_3,s_0) = 3.2$, $b_1 = 0.02$, $b_2 = 0.04$, $b_3 = 0.06$, $s_1 = 0$, $s_2 = 10$, $s_3 = 20$.

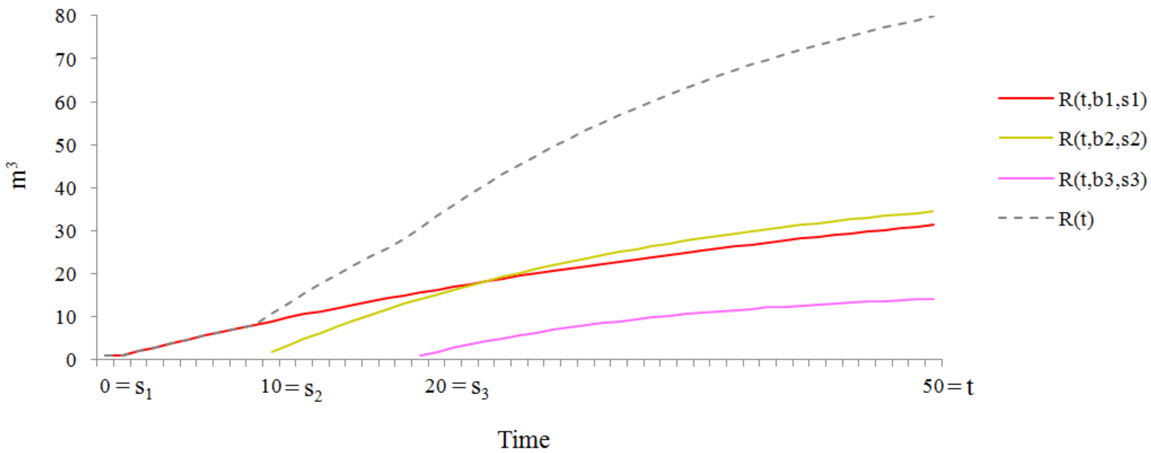
in Figure 3.3). The value of $G(t)$ decreases from $s_3 = 20$ on, as no more reforestations are undertaken.



Source: own elaboration

Figure 3.3. Growth of all vintages at time *t* when there are different types of species

The growth of trees belonging to each species accumulates over time, albeit differently from each other. In any case, the value of $R(t)$ is continually growing (see Figure 3.4).



Source: own elaboration

Figure 3.4. Accumulated past growth at time *t* when there are different types of species

The Social Planner's problem is now:

$$\max_{i(t),x(t),b(t)} \int_0^{\infty} U\left(C(t), A(R(t))\right) e^{-rt} dt \qquad [3.39]$$

subject to equations [3.2], [3.37], [3.38], the appropriate transversality conditions, and a modified version of [3.4]:

$$\dot{P}(t) = e(K(t)) - \eta \dot{R}(t) - \sigma P(t) \qquad [3.40]$$

The economy's total production at time t is now:

$$Y(K(t), x(t), b(t), P(t)) = C(t) + \phi(i(t)) + \varsigma(x(t)) \qquad [3.41]$$

Unlike previous scenarios, the time derivatives of [3.37] and [3.38] do not allow us to transform the problem into a standard optimal control model:

$$\dot{G}(t) = x(t) - \int_0^t b(s)x(s)e^{b(s)(s-t)} ds = x(t) - \int_0^t b(s)x(s)e^{-b(s)(t-s)} ds \qquad [3.42]$$

and

$$\dot{R}(t) = \int_0^t x(s)e^{b(s)(s-t)} ds = \int_0^t x(s)e^{-b(s)(t-s)} ds = G(t) \qquad [3.43]$$

where forest biomass at the given moment s continues to grow at a rate that depends on the type of species chosen at s and on the age-class structure (period of time between s and t). Thus, we can rewrite the Social Planner's problem as before, while replacing [3.37] by [3.42], [3.38] by [3.43] and [3.40] by

$$\dot{P}(t) = e(K(t)) - \eta G(t) - \sigma P(t) \qquad [3.44]$$

This problem needs to use the adaptation of the Hamiltonian[81] initially proposed by Vinokurov (1969) in order to solve it:

[81] Unlike previous scenarios, we now formulate the maximum principle by using the Hamiltonian, rather than the current-value Hamiltonian, as it makes the way of solving it easily. In any case, the results would have changed if we had chosen the current-value Hamiltonian, as explained below.

$$H(t)=U\left[C(t),A(R(t))\right]e^{-rt}+\lambda_1(t)\left[i(t)-\delta K(t)\right]+\lambda_2(t)G(t)$$
$$+\lambda_3(t)\left[e(K(t))-\eta G(t)-\sigma P(t)\right]+\lambda_4(t)x(t)-\int_t^\infty \lambda_4(s)b(t)x(t)e^{-b(t)(s-t)}ds \quad [3.45]$$

From the necessary conditions (see Appendix C), we also know that

$$\lambda_1(t)=p_i(t)e^{-rt} \quad [3.46]$$

$$\lambda_2(t)=\left(p_x(t)-p_{Fb}(t)+b(t)\frac{p_b(t)}{x(t)}\right)e^{-rt}+B_3(t)\eta e^{-rt} \quad [3.47]$$

$$\lambda_3(t)=B_3(t)e^{-rt} \quad [3.48]$$

$$\lambda_4(t)=\left(p_x(t)-p_{Fb}(t)+b(t)\frac{p_b(t)}{x(t)}\right)e^{-rt} \quad [3.49]$$

with $p_x(t)$ equal to [3.33] and:

$$p_{Fb}(t)=\frac{\partial Y(K(t),x(t),b(t),P(t))}{\partial x(t)}\,p(t) \quad [3.50]$$

$$p_b(t)=\frac{\partial Y(K(t),x(t),b(t),P(t))}{\partial b(t)}\,p(t) \quad [3.51]$$

$$B_3(t)=\left[\frac{(1+\delta)\dfrac{\partial\phi(i(t))}{\partial i(t)}-\dfrac{\partial Y(K(t),x(t),b(t),P(t))}{\partial K(t)}}{\dfrac{\partial e(K(t))}{\partial K(t)}}\right]p(t) \quad [3.52]$$

where $p_b(t)$ is the marginal revenue product of the species planted at time *t*.

After affine transformation and purging consumer surplus, the 'linearized' Hamiltonian is now (see Appendix C):

$$\hat{H}(t)=\left[p(t)C(t)+\frac{\partial U(t)}{\partial A(R(t))}A(R(t))+p_i(t)[i(t)-\delta K(t)]\right]e^{-rt}$$

$$+\left[\left(p_x(t)-p_{Fb}(t)+b(t)\frac{p_b(t)}{x(t)}\right)[G(t)+x(t)]+B_3(t)[e(K(t))-\sigma P(t)]\right]e^{-rt} \qquad [3.53]$$

$$-\int_t^{\infty}\left(p_x(s)-p_{Fb}(s)+b(s)\frac{p_b(s)}{x(s)}\right)e^{-rs}b(t)x(t)e^{-b(t)(s-t)}ds$$

As can be seen from [3.53], we have forward-looking values. We need future shadow values (not constant) to correctly evaluate the impact of current reforestations. The derivation of GNNP is now more complex than in previous scenarios as it is clear that final result maintains the integral which implies projections towards the future.

In short, we show that if

(III) there are various types of species each of them growing following a constant decay exponential growth function, as [3.36], but with a different values for the growth decay rate *b*,

then neither the ‘linearized’ Hamiltonian nor GNNP can be measured using values of the current year, as there are future values.

Note that in sections 3.2.1 and 3.2.2 we have assumed that trees are not felled. However, the possibility of felling trees after *T* years has been posed on the model developed in section 3.2.1 on a trial basis. In Appendix D we present this extension, although we have not been able to obtain reasonable expressions for the necessary conditions without the lambdas, so we cannot interpret them. One must take into account that the necessary conditions for solving this problem are different than those used in previous cases (Frankena, 1975; Kamien and Schwartz, 1991). In any case, it is shown that the current-value Hamiltonian depends on two points in time, i.e. that there are time-delayed variables. If we would apply these conditions, it could be expected that GNNP would also depend on two points in time.

In the case of the model presented in the section 3.2.2, if the problem got more complex and contained the possibility of felling, the result that the ‘linearized’ Hamiltonian and GNNP depend on future values (i.e. on infinite points) would be satisfied as well, as the fact that makes not possible to use values of the current year is the existence of different types of species that grow differently. We should also assume that each tree is felled at a different moment *T*, that is, that moment *T* varies depending on the species chosen. This assumption would also complicate the problem.

In a nutshell, values of the current year only give a good approximation of the real contribution of natural capital (forest biomass) to GNNP when there is a single species. Since this assumption is far-fetched, it seems more appropriate to measure forest capital in GNNP by discounting net future returns, as indeed the SEEA recommends (UN, 1993; UN et al., 2013, 2014). In Appendix E we address this issue in more detail.

3.3. Non-market ecosystem services

So far, the value of consumption of non-market *amenities* has been given by its contribution to the utility of individuals, $\frac{\partial U(t)}{\partial A(R(t))} A(R(t))$ (see [3.18]). However, in order to integrate non-market values into the national accounts in a consistent way with market values, we need to estimate exchange values for non-market goods and services. In doing so, we use the SEV method. Based on Caparrós (2010), and coming back to the benchmark model, let us suppose that the Social Planner decides to internalize forest recreation by establishing the obligation to make payments of an entrance fee to access the forest[82]. The number of times that the recreational service is rendered, i.e. the number of entrance fees purchased, is called υ .

The Social Planner's problem is now:

$$\max_{i(t),h(t),\upsilon(t)} \int_0^\infty U\left(C(t), A(\upsilon(t), R(t))\right) e^{-rt} dt \qquad [3.54]$$

subject to [3.2], [3.3], [3.4] and [3.5].

If we assume that $\upsilon(t)$ must actually be paid, the budget constraint [3.6] is given by:

$$Y(K(t), h(t), P(t)) = C(t) + \phi(i(t)) + \psi(h(t)) + \omega(\upsilon(t)) \qquad [3.55]$$

where $\omega(\upsilon(t))$ denotes the cost of the payment of the entrance fee.

[82] We refer to forest recreation for simplicity, but other non-market values could be equally treated.

The current-value Hamiltonian for the new problem is:

$$H_{c\_SEV}(t) = U[C(t), A(\upsilon(t), R(t))] + \lambda_1(t)[i(t) - \delta K(t)]$$
$$+ \lambda_2(t)[g(R(t)) - h(t)] + \lambda_3(t)[e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t)] \quad [3.56]$$

Apart from the necessary conditions derived in the section above ([3.10], [3.11] and [3.12]), we also obtain the following condition (see Appendix F):

$$\frac{\partial U(t)}{\partial C(t)} \frac{\partial \omega(\upsilon(t))}{\partial \upsilon(t)} = \frac{\partial U(t)}{\partial A(\upsilon(t), R(t))} \frac{\partial A(\upsilon(t), R(t))}{\partial \upsilon(t)} \quad [3.57]$$

where the Social Planner should equal the marginal cost in terms of the consumption of market goods and services to which individuals renounce due to the payment of the entrance fee (the L.H.S) and the change in consumption of non-market services that provides the payment of the entrance fee (which may be, for example, one-time access to the forest) multiplied by the variation in utility experienced by individuals who access the forest (the R.H.S). If we assume that $\frac{\partial A(\upsilon(t), R(t))}{\partial \upsilon(t)} = 1$, the 'quasi-market' price, $p_A(t)$, can be defined as the market value (in monetary units) of the entrance fee[83] such that:

$$\frac{\partial \omega(\upsilon(t))}{\partial \upsilon(t)} p(t) = \frac{\partial U(t)}{\partial A(\upsilon(t), R(t))} = p_A(t) \quad [3.58]$$

where the assumption $\frac{\partial A(\upsilon(t), R(t))}{\partial \upsilon(t)} = 1$ simply means that the payment of one additional entrance fee permits visitors an additional one-time recreational use of the forest.

According to the result obtained in [3.57], and applying the same necessary conditions as those laid down for the benchark model, i.e. [3.10], [3.11] and [3.12], we can rewrite the current-value Hamiltonian [3.56] as:

[83] $p_A(t)$ is not to be confused with $\upsilon(t)$. While $p_A(t)$ refers to the price of the entrance fee, $\upsilon(t)$ is the number of entrance fees purchased.

$$H_{c\_SEV}(t) = S[C(t), A(\upsilon(t), R(t))] + p(t)C(t) + p_A(t)[A(\upsilon(t), R(t))] \\ + p_i(t)[i(t) - \delta K(t)] + (p_w(t) - p_h(t))[g(R(t)) - h(t)] + B_1(t)[e(K(t)) - \sigma P(t)] \quad [3.59]$$

The definition of GNNP, this time in terms of market and 'quasi-market' values, implies once again that [3.59] must be linearized. Thus, we can write the following Remark.

Remark 1 *The 'linearized' current-value Hamiltonian [3.59] is equivalent to $GNNP_{SEV}$ estimated using exchange values. More precisely:*

$$\hat{H}_{c\_SEV}(t) = GNNP_{SEV}(t) = p(t)C(t) + p_A(t)[A(\upsilon(t), R(t))] \\ + p_i(t)[i(t) - \delta K(t)] + (p_w(t) - p_h(t))[g(R(t) - h(t))] \\ + B_4(t)[e(K(t)) - \sigma P(t)] \quad [3.60]$$

Remember that consumer surplus for both market and non-market goods and services have been removed when taking out $S[C(t), A(\upsilon(t), R(t))]$.

3.4. Exchange values and different market structures

Prices of market goods and services (i.e. those recorded in the SNA) are collected in $p(t)$ and calculated by balancing demand and supply functions in a competitive market. However, in real life, not all markets are perfectly competitive, so $p(t)$ also collects prices from monopolistic or oligopolistic markets. On the whole, there is a minimum price if perfect competition prevails and a maximum price if the market operates as a monopoly.

The SEV method proposes to utilize this information to simulate $p_A(t)$. This involves constructing demand and cost functions for these *amenities* and simulating at least the price that would be set up if perfect competition, monopoly or monopolistic competition prevails. This provides a range of values for $p_A(t)$. The exchange value for forest recreation is then calculated by multiplying $p_A(t)$ by the number of services consumed at that price, as explained in the second chapter. This valuation would not include consumer surplus[84].

3.4.1. Perfect competition

Under this assumption, the natural area is completely homogeneous with regard to other natural areas, so forest owners/managers cannot individually influence the price $p_A(t)$. Costs (essentially cleaning and warden costs) would also be covered with a standard return on invested capital. If we assume that the supply of the service requires an economically viable management system, at least fixed costs of the management must be covered by the 'quasi-market' price. This criterion was empirically applied in Caparrós et al. (2003) and Campos and Caparrós (2006).

Under these circumstances (existence of many forests, identical service and freedom to enter and leave the market), forest owners/managers only decide the optimal amount of recreational services to the offered at the given market price with the goal of maximizing profits. Thus, at this price, any 'sale' is possible. If entrance fees are sold above the market price, no one would be willing to pay for a service that may be offered by other forests under equal conditions. If we look at the opposite case, it would not be possible to meet the entire demand considering the size of the forest. This means that the site-specific demand function for recreational access is perfectly elastic, i.e. price equals marginal revenue.

Net profits, $\Pi^{pc}(t)$, are given by the difference between total revenue and total cost:

$$\Pi^{pc}(t) = p_A^{pc}(t)A^{pc}(\upsilon(t), R(t)) - c^{pc}\left(\omega(\upsilon(t)), \phi(i(t)), \psi(h(t)), A^{pc}(\upsilon(t), R(t))\right) \quad [3.61]$$

where *pc* is the superscript stand for perfect competition. The cost function can be redefined (for simplicity) as $c^{pc}(\upsilon(t), i(t), h(t), R(t))$ such that:

$$\Pi^{pc}(t) = p_A^{pc}(t)A^{pc}(\upsilon(t), R(t)) - c^{pc}(\upsilon(t), i(t), h(t), R(t)) \quad [3.62]$$

---

[84] The only way of internalizing consumer surplus into the market consists in assuming that the forest owner may charge each individual their maximum WTP, i.e. a different price to each visitor. This assumption is not realistic, so the premise of the SEV is that the forest owner may choose a unique price.

Profits are maximized at that point in which it is not possible to obtain any additional profit by allowing more visitors to access (application of the marginal analysis). This occurs when the *last* visitor accessing the forest provides exactly the same amount to total revenue as to total cost, that is, when marginal cost equals marginal revenue.

From the necessary conditions, we know that:

$$p_A^{pc}(t)\frac{\partial A^{pc}(\upsilon(t),R(t))}{\partial \upsilon(t)} = c^{pc'}(\upsilon(t),i(t),h(t),R(t)) \qquad [3.63]$$

As $\frac{\partial A^{pc}(\upsilon(t),R(t))}{\partial \upsilon(t)} = 1$, price also equals marginal cost.

In short, we confirm the rule on the basis of which a competitive forest must act in order to maximize profits: maximization of profits occurs at the point in which marginal revenue equals marginal cost. As forest owners/managers are price-takers, p_A^{pc} equals marginal revenue, so the maximization rule implies that p_A^{pc} also equals marginal cost.

3.4.2. Monopoly and monopolistic competition

Monopoly is a reasonable scenario if the forest has a relevant characteristic that makes it unique so that there are no perfect substitutes (this scenario was analyzed in Caparrós et al. (2003)). There is, nevertheless, a certain degree of similarity between monopolistic and perfect competition market structures when it comes to determining the level of visitors who may gain access to the forest. In both cases, the marginal revenue-marginal cost perspective is used. That is, profits are maximized by equating marginal revenue and marginal cost. If the former exceeds the latter, the forest owner/manager will permit more visitors to access the forest and vice versa. Conversely, the basic difference between a monopoly and a competitive forest is that price is not set by market forces in the former case. In other words, the demand curve of a forest owner/manager who acts as a monopolist is not horizontal, but rather downward-sloping.

Nevertheless, if there are a given number of different recreational natural areas which are to compete in the market, it is not reasonable to assume that all of them are either unique or homogeneous. Monopolistic competition represents an intermediate scenario (Caparrós et al., 2015). This market structure reflects reality better, at least in the case of forests. As product differentiation is possible, there are several forests offering close substitutes to each other. In any case, forests are not perfect substitutes, as they may have specific characteristics or may be located in different geographical places, among others specifications[85]. Thus, the forest owner has some pricing power, but this power is limited by the number of forests offering recreational activities.

In the short run the monopolistic competition model (Dixit and Stiglitz, 1977) is almost identical to the monopoly (i.e. profits are maximized following the same procedure as in a monopoly, so ordinary profits, extraordinary profits or losses can be obtained). New entries are, in principle, possible but need a considerable amount of time. In this way, the number of forests providing recreational services is fixed and each forest has a site-specific demand function. The equilibrium is given by the intersection between the site-specific marginal revenue function and the site-specific marginal cost function (Caparrós et al., 2015).

In the fourth and fifth chapters of this dissertation we define the revenue function for public forest recreation and a biodiversity conservation program as the price multiplied by the number of people who are willing to pay, which in turn depends on the price – i.e. $p^m(t)q^m(p^m(t))$–, where m is the superscript stand for monopolistic competition (this way of defining the revenue function is similar to that for the Bertrand competition in the context of oligopolistic firms). While it is more common to specify the revenue function in terms of quantities (as in the Cournot competition for an oligopoly), for these two particular services, it is more natural to think about forest owners/managers setting a price for accessing the forest and letting visitors decide whether or not they are willing to visit the recreational area at this price, rather than deciding the number of visitors that will have be allowed into the forest. This is even more evident for the case of a conservation program to protect the Iberian Lynx. In order to be consistent throughout the dissertation, we also define the revenue function in these terms and derive the results for a scenario in which there is product differentiation[86].

[85] If forests were perfect substitutes, visitors would choose the cheapest one. But as most visitors consider that forests are imperfect substitutes, the demand curve faced by each forest will be represented as downward-sloping.

[86] When forests are identical, the equilibrium of the Bertrand model is the competitive equilibrium.

Each forest owner/manager sets the price in order to maximize profits, so they compete in prices, thus leaving it up to the market to determine the number of visitors. Resulting from [3.58], it can be derived that when paying p_A one can obtain one entrance fee that gives the payer one-time recreational use of the forest. In this sense, $A(\upsilon(t), R(t))$ can be redefined as $A(p_A(t), R(t))$. This expression is more helpful under the market structure analyzed here, as the control variable for forest owners is now the price p_A, and not the number of entrance fees to be sold. The problem faced by the forest owner/manager is as follows[87]:

$$\max_{p_A(t)} \Pi^m\left(p_A^m(t)\right) = \left(p_A^m(t) - c^{m'}(t)\right) A^m\left(p_A^m(t), R(t)\right) \qquad [3.64]$$

From the necessary conditions, we know that:

$$A^m\left(p_A^m(t), R(t)\right) + p_A^m(t)\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)} = c^{m'}(t)\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)} \qquad [3.65]$$

so the forest owner/manager chooses a price such that marginal revenue equals marginal cost. From equation [3.65], we can derive the following reaction function:

$$p_A^m(t) = c^{m'}(t) - A^m\left(p_A^m(t), R(t)\right)\frac{1}{\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)}} \qquad [3.66]$$

Considering that an increase in the price brings about declining the number of visits, the quotient in the R.H.S of [3.66] is negative. This implies that the price $p_A^m(t)$ is higher than marginal cost, so the forest owner/manager obtains profits[88]. As we have just seen, marginal cost is equal to marginal revenue in the equilibrium, which means that the price $p_A^m(t)$ is also higher than marginal revenue. From equation [3.66]:

[87] In the applications of chapters four and five, we assume that all costs are fixed, as we have no information to simulate the variable component. We also make an adaptation to this formula to define the quantity of visits as the probability of paying the price (we shall come back to this issue).

[88] In so far as forests are imperfect substitutes, when one forest owner/manager reduces the price with respect to the others, not *all* visitors will choose the forest with a lower price. This implies that equilibrium prices will be above the competitive level.

$$p_A^m(t) = A^m\left(p_A^m(t), R(t)\right) + p_A^m(t)\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)} - A^m\left(p_A^m(t), R(t)\right)\frac{1}{\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)}}$$

so,

$$A^m\left(p_A^m(t), R(t)\right) + p_A^m(t)\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)} = p_A^m(t) + A^m\left(p_A^m(t), R(t)\right)\frac{1}{\frac{\partial A^m\left(p_A^m(t), R(t)\right)}{\partial p_A^m(t)}}$$

In short, we confirm the rule on the basis of which a forest under a monopolistic competition structure must act in the short run in order to maximize profits: maximization of profits occurs at the point in which marginal revenue equals marginal cost. As forest owners/managers are not price-takers, p_A^m is higher than marginal revenue, so the maximization rule implies that p_A^m is also higher than marginal cost.

If the demand function is linear and marginal cost is equal to zero, the price $p_A^m(t)$ that maximizes benefits coincides with the median, which facilitates the analysis. This scenario is perhaps an appropriate strategy for an application at the macro-economic level. Nevertheless, the assumption of zero marginal cost and the linearity of the demand function are not essential assumptions, as it is always possible to simulate the price that the forest owner would set. Caparrós et al. (2015) relax both assumptions and show that the monopolist (or the forest owner under a monopolistic competition scenario in the short run) would set a different price than the median, though relatively similar. In any event, the SEV method uses the value obtained under a monopolistic or monopolistic competition position as the maximum price of the visit. This is a situation that yields extraordinary profits to the monopolist or to the forest owner under a monopolistic competition situation (he/she collects profits above the opportunity cost in competitive capital markets).

In the long-term new forests offering recreation activities are supposed to enter the market in the original model, although each forest operates with its own site-specific demand function (hence monopolistic competition and monopoly no longer coincide under this scenario). This mechanism operates until economic profits cease to exist. This occurs when $p_A^m(t)$ equals average cost, *AC*, which is not zero in case of fixed cost, even if marginal cost is zero. In an equilibrium state, the price is higher than the respective marginal cost, however.

Whether the short-term pricing policy or the long-term pricing policy is more reasonable to obtain an exchange value depends on how credible is the 'introduction of new recreational forests in one particular area'. Most applied studies analyze market structures in the short-term, as they are focused on a site-specific analysis. It is in this spirit that we present the fourth chapter of this dissertation. Although the type of forests analyzed, as well as the type of competition they face, may change in the future, we are only interested in estimating the value of public recreation in the forests that currently exist and the competitive environment within which they operate at present. Furthermore, national accounting systems also recommend focusing on the short run.

3.5. Intermediate summary

This chapter sets out the main findings for the forestry sector obtained by the theoretical literature on green national accounting. From this perspective, we address the valuation of non-market ecosystem services and the treatment of future values.

As regards the former, we show that non-market ecosystem services can be integrated into the national accounts at their (*simulated*) exchange values. We have provided theoretical support for the SEV method, which has been addressed from a green national accounting perspective. Rather than using prices for similar markets to include the final *output* of the forests, this method simulates the 'quasi-market' price that would be set if these services were internalized in the market. In principle, nothing distinguishes a service like public forest recreation (presently outside of the market as access is free, but which could be incorporated) from a non-timber product like mushroom picking in Sweden (also presently outside of the market as picking is free) (Campos and Caparrós, 2006). Thus, this method has proven to be a methodology in line with the central normative procedure of the SNA and the SEEA.

As regards the treatment of future values when evaluating the stock of forest biomass, previous studies on forests (see, e.g. Cairns (2001, 2003)) assume that forests grow instantaneously. This allows defining the current-value Hamiltonian and GNNP in terms of values of the current year. However, we show that this result is only fulfilled under certain restrictive assumptions, which imply using a single species. When there are different species growing differently, there are forward- looking values and the Hamiltonian and GNNP include a Volterra integral equation that cannot be simplified to the standard state equation by taking the time derivative. This type of equations had not been applied so far within a green accounting approach, so we make a useful contribution in this regard, by extending conventional ‘vintage’ models to the modeling of the growth of forest biomass.

Moreover, as explained in the previous chapter, central normative national accounts frameworks propose to use the NPV approach in order to estimate the value of ecosystem stocks (in our case, forest biomass). NPV calculates the discounted present value of expected future returns (UN, 1993; UN et al., 2013, 2014), which requires making projections into the future.

CHAPTER 4. AN APPLICATION TO PUBLIC RECREATION IN SPANISH FORESTS

4. AN APPLICATION TO PUBLIC RECREATION IN SPANISH FORESTS[89]

As explained in the second chapter, in the last three decades there has been an effort to develop valuation techniques intended to integrate non-market benefits from ecosystems into economic analyses. Among these, public (free access) forest recreation is one of the most extensively studied subjects (Scarpa et al., 2000; Christie et al., 2007; Huhtala and Pouta, 2008; Rosenberger et al., 2012; Abildrup et al., 2013; Saelen and Ericson, 2013). Contingent valuation was originally the most widely used of all these methods, but choice experiments have attracted attention as an alternative due to the advantages associated with multi-attribute valuation. Although these methods are mainly applied to extended CBA, which uses Hicksian variation measures, there is an increasing interest in their use for national and ecosystem accounting, which require exchange values (Campos and Caparrós, 2016; Obst et al., 2016).

In this chapter we present the results of a choice experiment applied to the valuation of public recreation in stone pine (*Pinus pinea*) and cork oak (*Quercus suber*) forests in Spain. Based on this choice experiment, we compare the compensating variation measure (a type of Hicksian variation) associated with a visit to these forests with the results obtained from the SEV method for the same visits (Caparrós et al., 2003, 2015). The latter method allows us to estimate the potential benefits that could be obtained from internalizing this service in a potential market of recreation in these forests.

As explained in section 2.2, estimating these two measures relies heavily on the validity of the non-market valuation method used and the application of choice experiment is questioned due to its hypothetical nature (this also applies to contingent valuation) (Hausman, 2012). In this context, convergent validity tests offer ways to validate the results from these methods by assessing whether different techniques,

[89] This chapter is based on Oviedo et al. (2016).

formats and/or characteristic of the valuation scenario converge to similar WTP estimates (Hausman, 1993)[90]. Thus, and previous to the estimation of compensating variation and simulated exchange value measures, we empirically test the convergent validity of two elicitation formats and two payment vehicles in our CE application with the aim of validating the results from the experiment.

4.1. Background

Convergent validity of elicitation formats in CE mostly have compared rating, ranking and choice, although CE practitioners tend to prefer ordinal measures (ranking and choice) (Roe et al., 1996). Previous comparisons have given special attention to choice and ranking recoded as a choice formats, showing divergent results in earlier studies (Boyle et al., 2001; Mogas and Riera, 2001) and convergent validity in later studies (Caparrós et al., 2008; Akaichi et al., 2013). These comparisons, except Boyle et al. (2001)[91], are performed in an experiment with two alternatives plus the status quo. In this chapter, we extend this by comparing the results from a choice and a ranking recoded as a choice in a split-sample design experiment with three alternatives plus the status quo, which implies additional information and complexity in the choice task compared to previous studies.

Payment vehicles have also been compared in split-sample designs of choice experiment (Swallow and McGonagle, 2006; Biénabe and Hearne, 2006; Nunes and Travisi, 2009; Kaczan et al., 2013). In the procedure, a question including a different payment vehicle is randomly assigned to each respondent. These studies show that WTP estimates are statistically different across payment vehicles, highlighting the need for

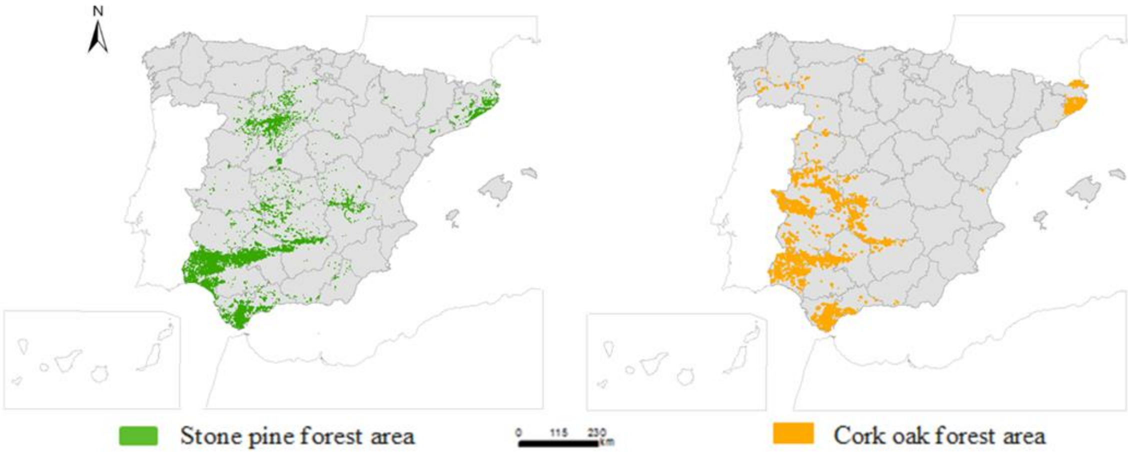
[90] See, for example, Cameron (1992); Adamowicz et al. (1994); Whitehead et al. (1995); Huang et al. (1997); Whitehead et al. (1998); Boyle et al. (2001); Mogas and Riera (2001); Azevedo et al. (2003); Scarpa et al. (2003); Dosman and Adamowicz (2006); Eom and Larson (2006); Caparrós et al. (2008); Azevedo et al. (2009); Christie and Azevedo (2009); Whitehead et al. (2010); Hoyos and Riera (2013); Akaichi et al. (2013); Jeon and Herriges (2016).

[91] Although Boyle et al. (2001) used three alternatives plus the status quo, their experimental design was random in attributes, implying that the complete status quo alternative appeared only in some of the choice sets. This design is not usual in choice experiment and has the drawback that not having the status quo alternative in all choice sets makes it difficult to obtain adequate welfare measures (Roe et al., 1996). However, it may be case that it would be reasonable to ignore the status quo alternative, as there is no possibility of opting out, for example, for a legal obligation (Olschewski et al., 2012; Olschewski, 2013).

further exploration of these divergences in choice experiment applications. In our choice experiment we compare two payment vehicles: an entrance fee and increased trip expenditures (due to increased gas prices), by including them in the same alternatives presented in the choice set. This particular design is inspired by previous hunting valuation studies that include both hunting trip costs and hunting fees as attributes in the same alternative (e.g., Mackenzie (1993)). We intend to identify how respondents make trade-offs between alternatives with different cost levels for the two payment vehicles presented. Thus, we perform a convergent validity test that analyzes whether the parameters (utility weights) of the different payment vehicles in the same utility function are similar or not; that is, whether the marginal utility of money associated with each payment is similar or not. In principle these utility weights should be similar, as both payments have the same consequence: reducing respondent's income. However, Campos et al. (2007) find the opposite for the same payment vehicles that we analyze and in a similar empirical context, although they use a split-sample design and the contingent valuation method. They also show that the entrance fee generates a significantly higher proportion of protest response. Our design extends this analysis in a choice experiment setting, where trade-offs between alternatives are more explicit. This design also tries to minimize the protest response effect associated with the entrance fee.

4.2. Survey and design

Stone pine and cork oak trees are both characteristic of the Mediterranean landscape. They are found mostly on the Iberian Peninsula and in other countries in the Mediterranean basin. The cork oak tree has a thick, insulating bark that offers the raw material used for making cork bottle stoppers. The stone pine produces an edible seed, the pine nut, which is highly valued in the food sector. Firewood is also obtained as a by-product of silvicultural practices. In addition, cork oak and stone pine forests provide non-market services; e.g., preservation of biodiversity, landscape sightseeing, carbon sequestration and public recreation (Caparrós et al., 2010) and disservices, e.g., plantations increase evapotranspiration. In Spain, the forests where these species are dominant are located mainly in the southwest, west, northwest and northeast areas (see Figure 4.1); although there are important differences between the first three areas and the fourth.



Source: own elaboration based on MAGRAMA (2014a and 2014b)

Figure 4.1. Distribution of stone pine (*Pinus pinea*) and cork oak (*Quercus suber*) forest area in Spain

In the southwest, west and northwest these forests are managed open woodlands with low tree density. Trees are removed to increase the growth of grass and cork in cork oak forests and to foste pine-cone production in stone pine forests. This allows for easy access to these forests, which are usually found in contiguous extensions and large properties, although stone pine stands are also found in small pine groves. The cork oak landscape features an open woodland with scattered trees on hills with gentle slopes. The understory is mainly comprised of grasslands, which provide grazing for livestock and game. Cork oak stands can be pure or mixed with other tree species, such as holm oak (*Quercus ilex*), wild olive (*Olea sylvestris*) and stone pine. In the understory, we also find shrub species such as mastic (*Pistacia lentiscus*) and rockrose (*Cistus spp.*). The main characteristic of the stone pine tree is its singular umbrella shape. Grazing on stone pine forests is less common than in cork oak forests because of lower grass productivity. Stone pine forests can also be pure or mixed with other tree species such as holm oak, cork oak and Aleppo pine (*Pinus halepensis*). They either have no understory vegetation or may be accompanied by coppices of oaks and by other Mediterranean shrubs such as rockrose.

In northeastern Spain, stone pine and cork oak forests are characterized by being fragmented (properties are small), minimally managed, densely forested and difficult to access. Commercial production of cork and pine nuts are also found in some areas, but there are many properties where this type of production has been abandoned. The result is a forest that usually functions as a wild garden for the cottage houses (known as *masias*) that landowners have on their properties. These characteristics determine a different level of access to and provision of public recreation services than in the forests of the southwest, west and northwest areas.

Our choice experiment targets recreation carried out by public visitors in the stone pine and cork oak forests located in the southwest, west and northwest of Spain, as they are more similar to each other and more suited for public recreation than forests in the northeast. In addition, cork oak forests in the southwest, west and northwest areas represent 92% of the cork oak forest area in Spain, while stone pine forests in these regions represent 86% of the stone pine forest area in Spain[92].

We included the choice experiment in a survey that was administered to a sample of 604 Spanish adults (>18 years old) from 14 provinces located in the southwest, west and northwest of Spain (*Cádiz, Málaga, Sevilla, Córdoba, Huelva, Badajoz, Cáceres, Valladolid, Madrid, Segovia, Toledo, Salamanca, Zamora and Ávila*). We selected these provinces to draw our sample as they contain or are adjacent to regions that contain stone pine and cork oak forests, so most respondents know about or are familiar with these forests. We stratified the sample by provinces, based on population and randomly selected within each province. A professional company interviewed individuals from the sample at their homes from May to July 2008. The survey response rate was 70%.

Although our initial sample was 604 respondents, the choice experiment was designed to be presented only to forest recreationists. Individuals not making an active use of these services are out of the scope of our valuation exercise. Thus, the valuation scenario started by asking respondents if they had made a recreation visit to a forest in Spain at least once in the past 12 months. Out of our initial sample of 604 individuals, 336 answered affirmatively and were qualified as forest recreationists[93]. These respondents represent our final sample for the choice experiment.

The choice experiment asked respondents first to think about the next visit they were planning to a forest and to imagine that, eventually, the expenses they incurred

[92] For brevity, when we refer to stone pine and cork oak forest in Spain throughout the chapter, we refer to forests located in the southwest, west and northwest areas.

[93] Respondents who declared no participation in forest recreation skipped the CE. Previously, these respondents faced a series of attitudinal questions and a valuation scenario about the preservation of the Iberian Lynx. The analysis of these questions is presented in chapter 5.

during that visit increased because both the forest owner established an entrance fee for accessing the forest and there was an increase in trip expenditures (we asked respondents to think on a potential increase in these expenses but not to state a monetary value for this increase). This created the context for the valuation in the subsequent choice task. Respondents then faced two choice sets including three forest visit alternatives plus the status quo. The status quo implied not doing any of the visit alternatives presented in the choice set. Half of the sample had to choose one alternative and half of the sample had to rank the alternatives from most to least preferred.

The forest visit alternatives were characterized by the following attributes and levels (see Table 4.1): the type of forest visited (*TREE*), the presence of recreation infrastructures (*INS*) (e.g., picnic tables, visitors' centers, trail information), the possibility of seeing domesticated animals (*ANI*), the possibility of picking mushrooms (*MUSH*), the entrance fee (*ENT*) and the increased trip expenditures (*EXP*). We used two focus groups and a pre-test to identify the main attributes of a forest recreation visit, to design the vector of monetary values offered in the CE and to evaluate a preliminary version of the survey. We completed the pre-test with 50 individuals[94]. The pre-tests tested a preliminary set of attributes, where the presence of infrastructures for recreation and the possibility of sightseeing animals were highly scored. The possibility of picking mushrooms also appeared to be a valued attribute. This is probably because mushroom picking in some Spanish forests (particularly in those in the southern region of Andalucía) has a main recreation component independent of whether or not the harvested mushrooms are sold. Martínez Peña et al. (2015) show that only 4% of mushroom collectors in the forest of Andalucía have a commercial purpose while the remaining 96% are recreation-oriented.

Table 4.1. Attributes of the Experiment and Levels

Attributes	Levels
Type of forest (*TREE*)	Stone pine; cork oak
Infrastructures for recreation (*INS*)	No; Yes
Domestic animals (*ANI*)	No; Yes
Mushroom collecting (*MUSH*)	No; Yes
Entrance fee (*ENT*)	€2; €7; €12; €17
Increased trip expenditures (*EXP*)	€2; €7; €12; €17

[94] Individuals participating in the pre-test and in the focus groups were general public and not only forest recreationists.

Using the universe of 256 combinations ($4^2 \times 2^4$) of attribute levels (Table 4.1), we obtained 16 alternatives that form a main effects design. We then combined these alternatives in sets of three alternatives (plus the status quo), obtaining 16 choice sets that form an orthogonal design of combination of alternatives with an efficiency of 100%. This design was also balanced in attribute levels. To reduce the cognitive effort of respondents, we blocked these 16 choice sets in eight questionnaire types with two choice sets each. To avoid potential order-effects, we replicated these questionnaires but changed the order of presentation of the choice sets. Thus, we ended up with 16 questionnaire types presenting two choice sets each. The questionnaire type was randomly assigned to each individual. Appendix H presents an example of a choice and ranking set.

Our sample is split up into 174 respondents facing the choice format and 162 respondents facing the ranking format. We identified 10 protest responses in the choice sample (6% of the sample) and 12 protest responses in the ranking sample (10% of the sample). A chi-square test shows no significant difference between these protest response rates (χ^2 statistic = 2.646). We identified protest responses by asking those who chose the status quo in both choice sets the reasons for their decision. When the answer to this question showed a clear rejection of the scenario presented (e.g., 'I already pay enough taxes' or 'We have the right to freely access these forests') we classified the respondent as a protest. For the remaining sample, we identified 2 respondents in the choice sample (1% of the sample) and 7 respondents in the ranking sample (5% of the sample) that did not state the number of people they pay for during their forest visits. Again, a chi-square test shows no significant difference between these rates of non-response (χ^2 statistic = 0.099). As this information is necessary to analyze the choice data (see section 4.3), our final sample is 162 respondents in the choice sample and 143 respondents in the ranking sample (324 and 286 observations, respectively, as each respondent faced two choice sets).

4.3. Analysis

We present now the econometric specification of the models for the choice and the ranking recoded as a choice (hereafter recoded choice) responses, the comparison tests used and the formulas and calculations needed for estimating aggregated economic values for forest recreation.

4.3.1. Econometric models

We assume a linear-in-parameters utility function for individual *l* and alternative *j* in a set of *J* alternatives (*j = 1, 2, 3, 4*) and for choice situation *u* (*u = 1, 2*) with a systematic (V_{lju}) and a random component (ε_{lju}):

$$U_{lju} = V_{lju} + \varepsilon_{lju} = \beta' X_{lju} + \varepsilon_{lju} \qquad [4.1]$$

where β represents a vector of parameters, X_{lju} is a vector of observed variables for alternative *j*, individual *l* and choice situation *u*; and ε_{lju} are random errors. We include as explanatory variables in the models the non-monetary attributes from Table 4.1 (*TREE, INS, ANI, MUSH*) plus a constant specific to forest visit alternatives (*ASC-REC*). The two monetary attributes from Table 4.1, which represent the total additional payments that the respondents would incur on the visit to the forest, are also included in the models but they are recoded to per-person values. Participants in the focus groups stated that it was easier to think in terms of total additional payment. However, as our goal is to estimate the WTP per visit per person, we asked respondents to indicate in the survey the number of people they usually pay for during their visits. Based on this, we recode the monetary attribute values to euros per person (named now *ENTP* and *EXPP*) to be included in the models.

Given the previously defined utility function, the probability that the respondent *l* chooses alternative *j* over any alternative *y* ($\forall y \in J$) in choice situation *u* ($\Pr_{lju}$) is:

$$\Pr_{lju} = \Pr\left[V_{lju} + \varepsilon_{lju} > V_{lyu} + \varepsilon_{lyu}\right] = \Pr\left[V_{lju} - V_{lyu} > \varepsilon_{lyu} - \varepsilon_{lju}\right] \forall j, y \in J \qquad [4.2]$$

Different assumptions about the density function of random terms $f\left(\varepsilon_{lju}\right)$ give different models. We start with a baseline model, the conditional logit, and further develop our analysis with the mixed logit model (Train, 2009). In the conditional logit, errors are assumed to be independently and identically distributed with an extreme value distribution across the *h* alternatives ($\forall y \in J$), *l* respondents and *u* choice situations. This gives the following probability:

$$\Pr_{lju} = \frac{e^{\mu\beta' X_{lju}}}{\sum_{y \in J} e^{\mu\beta' X_{lyu}}} \qquad [4.3]$$

where μ is the scale factor, which is normalized to 1. This distribution implies that the ratio of the probabilities of choosing any two alternatives is independent of the remaining alternatives. This is known as the independence of irrelevant alternatives (IIA) hypothesis. The violation of this assumption may arise when some alternatives are qualitatively similar to others, as occurs in our experiment. Concerning the repeated choice situations faced by each respondent, we assume that they are independent; that is, we treat each choice situation as a different observation.

The mixed logit is constructed on the assumption that the attribute parameters now have a component common to all individuals and an individual specific component. The utility function takes the following form:

$$U_{lju} = \beta_l^{'} X_{lju} + \varepsilon_{lju} \quad [4.4]$$

where X_{lju} is a vector of variables for alternative j, individual l and choice situation u, β_l is a vector of parameters for person l (the individual specific component) and ε_{lju} are random errors that are independently and identically distributed with an extreme value distribution (the common component). The individual specific component follows a distribution with density $f(\beta)$ which is a function of parameters θ (for example, the mean and covariance of β in the population). This distribution is specified by the researcher. This model has the advantage of allowing correlated error terms between alternatives and between repeated choices made by each respondent, not assuming the IIA hypothesis and modeling unobservable heterogeneous preferences (Train, 2009).

As this mixed logit model takes into account repeated choices by each respondent, the probability of a sequence of choices is the integral of the product of the conditional logit probabilities in [4.3] over all possible values of β_l according to $f(\beta|\theta)$ (Train, 2009). This integral has no closed-form solution but it can be evaluated through simulation for any value of parameters θ. Being Ψ the number of draws from θ (in our models $\Psi = 500$), the unbiased estimator of $\Pr_{lju}$ in the mixed logit is defined as (Train, 2009):

$$\breve{\Pr}_{lju} = \frac{1}{\Psi} \sum_{r=1}^{R} \left[\prod_{t=1}^{T} \frac{e^{\mu \beta_l^{r'} X_{lju}}}{\sum_{y \in J} e^{\mu \beta_l^{r'} X_{lyu}}} \right] \quad [4.5]$$

In this application, we assume a normal distribution for random parameters, except for the payment attributes and the *ASC-REC*, which we assume to have fixed parameters[95].

The payment attributes (*ENTP* and *EXPP*) are coded as continuous variables (in euro values). The attributes *INS*, *ANI* and *MUSH* are dummy-coded (1 for the presence of the attribute in the alternative and 0 otherwise). The attribute *TREE* is effect-coded (-1 for stone pine and 1 for cork oak) to differentiate the effect of choosing any of the two possible forest visit alternatives from the effect of choosing the status quo. The *ASC-REC* is dummy-coded (1 for forest visit alternatives and 0 otherwise). The status quo attribute levels are normalized to zero. We used NLOGIT version 4.0 for estimating the parameters through maximum likelihood for the conditional logit and through simulated maximum likelihood for the mixed logit.

From the models we generate empirical distributions for the individual parameters of each *k* attribute through the Krinsky and Robb (1986) bootstrapping technique. We take 1,000 random draws from the distribution of the parameters and their corresponding variance-covariance matrix. Then we use the resulting empirical distributions to estimate the mean marginal WTP for each *k* attribute using the formula $-\beta_k / \beta_{ENTP}$ (or $-\beta_k / \beta_{EXPP}$ depending on the payment vehicle used in the calculation) and the mean WTP for a recreation visit to a forest given the values of the attributes characterizing that specific visit. The standard deviation and the 95% confidence interval of these estimates are calculated using the percentile approach (Efron and Tibshirani, 1993).

4.3.2. Comparison tests

To find out whether the choice and the recoded choice models provide similar parameter vectors we use a Likelihood Ratio test. We follow the proposal by Swait and Louviere (1993) that makes it possible to test whether divergences are due to differences in taste or differences in scale parameters.

[95] Specifying fixed parameters for the payment attributes allows for the WTP of each attribute to follow the same distribution as its random parameter (Revelt and Train, 1998). The option of specifying a normal distribution for the parameters of the payment attributes could imply behaviorally inconsistent WTP values. Alternatively, setting a different distribution for these parameters makes the interpretation of WTP values more difficult.

Our null hypothesis is:

$$H_A : (\mu_{CH}\beta_{CH}) = (\mu_{RC}\beta_{RC}) \quad [4.6]$$

where CH refers to choice and RC refers to recoded choice. To falsify this hypothesis, the test separately examines two hypotheses. In hypothesis:

$$H_{A1} : (\beta_{CH}) = (\beta_{RC}) \quad [4.7]$$

where the relative scale parameter is set as μ_{CH} / μ_{RC}. If H_{A1} is rejected, H_A is also rejected and differences derive from taste parameters. If H_{A1} is not rejected, we test hypothesis:

$$H_{A2} : (\mu_{CH}) = (\mu_{RC}) \quad [4.8]$$

where the scale parameters are constrained to be equal under the null hypothesis. If H_{A2} is rejected, H_A is also rejected and the differences derive from scale parameters. If both H_{A1} and H_{A2} are not rejected, then H_A is not rejected.

We also compare the mean marginal WTP measures obtained with each format by applying the Krinsky and Robb (1986) bootstrapping technique. We do this for the marginal mean WTP estimated for attributes *ASC-REC, TREE, INS, ANI* and *MUSH* and for the mean WTP obtained for a visit to either a stone pine or a cork oak forest with no additional attributes. As we use two payment vehicles, we estimate separate WTP values for *ENTP* and *EXPP*; that is, in each case we divide the β parameters of the attributes by the β parameter of either *ENTP* or *EXPP*. Then, using the complete combinatorial test (Poe et al., 2005) we test the null hypothesis:

$$H_B : (WTP_{CH,k,pv}) = (WTP_{RC,k,pv}) \quad [4.9]$$

where k corresponds either to an attribute or to a forest visit and pv corresponds to the payment vehicle used.

To test differences between payment vehicles, we use a *t-test* to compare the parameters of the attributes *ENTP* and *EXPP*. Our null hypothesis here is:

$$H_C : (\beta_{ENTP}) = (\beta_{EXPP}) \quad [4.10]$$

We also use the complete combinatorial test to compare the WTP obtained with each payment vehicle for each attribute and forest visits defined previously (a visit to either a stone pine or a cork oak forest with no additional attributes). We test the null hypothesis:

$$H_D : \left(WTP_{ENTP,k}\right) = \left(WTP_{EXPP,k}\right) \quad [4.11]$$

where *k* corresponds either to an attribute or to a forest visit.

4.3.3. Estimation of aggregated values

The Compensating Variation (*CV*) offers, given a specific set of attribute values, the welfare measure associated with the provision of the public recreation services of a forest visit. This is of interest for CBA and when using a welfare maximization approach. According to Small and Rosen (1981), the *CV* for forest alternative *j* from the conditional logit (CV_j) is:

$$CV_j = \frac{1}{\beta_{pv}}\left[\left(\beta' x_0\right)^{NM} - \left(\beta' x_j\right)^{NM}\right] \quad [4.12]$$

where $\left(\beta' x_0\right)^{NM}$ and $\left(\beta' x_j\right)^{NM}$ are the part of the utility corresponding to the non-monetary attributes in alternatives *0* (the status quo) and *j* (a forest visit alternative) respectively[96]. β_{pv} is the parameter of the payment vehicle used in the calculation. Alternative *j* represents visiting either a stone pine or a cork oak forest with no additional attributes. We consider this to be the most common scenario of a recreation visit in the analyzed stone pine and cork oak forests. When using the mixed logit model, the *CV* for forest alternative *j*, CV_j^E, is:

$$CV_j^E = \frac{1}{\beta_{pv}} E\left(\left[\left(\beta' x_0\right)^{NM} - \left(\beta' x_j\right)^{NM}\right]\right) \quad [4.13]$$

where the expectation, E, is taken with respect to the random parameters. To calculate this, we use stochastic simulation as in the mixed logit model in [4.5], estimating CV_j 1,000 times where CV_j^E is the mean of these 1,000 estimates. The aggregated CV_j (or CV_j^E) in a year is estimated multiplying CV_j (or CV_j^E) by the annual number of visits to the forest *j*, Q_j.

[96] We drop subscript *u* as this estimation would be the same regardless of the choice situation faced by the respondent.

We also present the SEV measure, which offers the benefits that could potentially be collected in a real market for a non-market service. As explained in the previous chapter, this approach assumes that the forest owner/manager would set a single price for the provision of forest recreation services with the goal of maximizing benefits. Therefore, only part of the population would pay that price in the case that the non-market services were internalized. To estimate this price, the method uses a demand function estimated with non-market valuation techniques (a choice experiment in our case) and a supply function based on the commercial costs associated with the provision of the service. In our application we assume that all costs incurred in the provision of recreation services are fixed because we have no information to simulate a cost function with a variable component. Under this assumption calculations are simpler, as the maximization of benefits occurs at the same price as the maximization of revenues.

As explained in the third chapter, we calculate the revenues from a visit to forest alternative j, R_j[97], in the hypothetical market as the price set for accessing a forest j, p_j, (the increased payment respect to the current situation)[98] multiplied by the quantity of visits to a forest j, q_j, which depends on p_j:

$$R_j = p_j \cdot q_j(p_j) \qquad [4.14]$$

Assuming that at a zero price the number of visits equals the current visits under free access (i.e. $q_j(0) = Q_j$), we express the quantity of visits as the probability of paying a price for visiting a forest j, Pr_j, multiplied by the current number of visits to a forest j, Q_j:

$$q_j(p_j) = \mathrm{Pr}_j(p_j) \cdot Q_j \qquad [4.15]$$

Therefore, we express the revenue function as:

$$R_j = p_j \, \mathrm{Pr}_j(p_j) \cdot Q_j \qquad [4.16]$$

[97] In previous chapter *R* represents stock of renewable resources, buy we now use this notation to define the revenue function.

[98] This increased payment can take any form that allows collecting the money in the hypothetical market. Any mechanism that allows charging p_j applies here.

Given that the increased payment in $\Pr_j$ is equivalent the price to be paid for the visit to a forest *j*, p_j, we calculate the first derivative of R_j in respect to p_j and find the p_j for which the first order condition of this derivative holds. To estimate q_j, we substitute p_j in [4.15], using the probability function from the estimated model. We use the estimated p_j and q_j to calculate the maximum revenue from a visit to a forest *j* (R_j), which is the SEV of public recreation.

We perform this simulation for the next recreation visit that the forest visitors would make and then we assume that the estimated values are similar for the additional visits that these visitors make in the year. This implies that we simulate a single choice situation (*t*) in the probability function. This has no impact when we use the conditional logit, as this model assumes that choice situations are independent (even those faced by the same respondent) and the probability function is always [4.3]. In the case of the mixed logit, this simplifies the probability function because when $u = 1$ the product of conditional logit probabilities in [4.5] includes a single probability[99].

We estimate q_j, p_j and R_j in two scenarios:

(i) One alternative of a forest visit plus the status quo (*SEV-2*)

(ii) Two alternatives of a forest visit plus the status quo (*SEV-3*)

The second scenario has the advantage of incorporating substitutes into the decision of visiting a forest. In both scenarios alternative *j* corresponds to visiting either a stone pine forest or a cork oak forest with no additional attributes.

In *SEV-2*, we find p_j^* (see Appendix I) when:

$$e^{(\beta' x_j)} + 1 + (\beta_{pv} \cdot p_j) = 0 \qquad [4.17]$$

[99] Ideally, the choice experiment design should have included in the questionnaire as many choice situations as number of annual recreation visit to forests made by the respondent. Unfortunately, this information was not available *a priori*.

As this equation has no analytical solution, we have to obtain p_j by iteration. For the mixed logit model, we estimate p_j^* by using stochastic simulation as in equation [4.5] when:

$$E\left(e^{(\beta' x_j)}\right) + 1 + \left(\beta_{pv} \cdot p_j\right) = 0 \qquad [4.18]$$

We take the expectation, E, with respect to the random parameters; we estimate p_j 1,000 times and we use the mean of p_j to calculate q_j and then R_j.

In *SEV-3*, we assume that the same price would be set in each forest alternative. In this case, we find p_j (see Appendix I) when:

$$e^{(\beta' x_j)} + e^{(\beta' x_y)} + 1 + \left(\beta_{pv} \cdot p_j\right) \qquad [4.19]$$

As in *SEV-2*, we find p_j by iteration. For the mixed logit, we find p_j by using stochastic simulation when:

$$E\left(e^{(\beta' x_j)}\right) + E\left(e^{(\beta' x_y)}\right) + 1 + \left(\beta_{pv} \cdot p_j\right) = 0 \qquad [4.20]$$

As in *SEV-2* we take the expectation, *E*, with respect to the random parameters; we estimate p_j 1,000 times and we use the mean of p_j to calculate q_j and then R_j.

4.4. Results

4.4.1. Socioeconomic characteristics of the sample

The average forest recreationist in our sample is a middle-aged person, with a net family income of almost €2,000 per month, a family of three members, a similar chance to be either a man or a woman, more likely to live in an urban center and more likely to have a medium level of education (between a high school and college degree). This person spends on average almost €9 per person and day during a visit to a forest (Table 4.2). While there are no official data available that offer information on the characteristics of forest recreationists in Spain, we can contrast our sample characteristics with those from previous studies on public recreation in Spanish forests (Arriaza et al., 2002; Campos et al., 2007; Samos Juarez and Bernabeu Cañete, 2011; Martínez-Carrasco et al., 2012). The characteristics from our sample are within the range of values offered by these studies except for gender; our sample offers a slightly higher percentage of women. In previous studies age is between 33 and 43 years, net family income is between €1,718 and €2,353 (updated to 2008 Euros), family members are between 3 and 4, percentage of woman is between 41 and 43 and trip cost per person and day is between €7.87 and €20.02 (updated to 2008 Euros). Education shows average levels between medium (high school) and high (college). No study offered information about percentage of respondents living in an urban center.

Table 4.2. Socioeconomic characteristics of the forest recreationists facing the choice experiment

Variables	Choice experiment sample	
	Mean	n
Age	36 (13)	300
Net family income (€ per month)	1,992 (918)	259
Family members	3.2 (1.3)	299
Gender (1=woman; 0=man)	0.47 (0.50)	305
Education (1= college degree or more; 0=otherwise)	0.39 (0.49)	305
Urban centre (% of respondents living in city with population larger than 20,000 inhabitants)	0.68 (0.47)	305
Trip cost per person per day (€)	8.71 (11.90)	281

Note: standard errors are shown in parenthesis; n: number of observations.

4.4.2. Choice versus recoded choice

Mean parameters from the models[100] for both the choice and the recoded choice samples are all significant, with the exception of *TREE* in the recoded choice sample (Table 4.3). The *ASC-REC* shows a positive sign, which indicates a preference for some of the presented forest visit alternatives. The negative sign of *TREE* shows a preference for visiting stone pine forests versus cork oak forests. This result is consistent with the fact that stone pine forests receive more visitors in Spain than cork oak forests, both in total and on a per hectare basis (see section 4.3). This may be related to a major proximity of

[100] Chi-square statistics for the IIA tests when removing the first, second and third alternative are 8.347, 23.620***, and 20.067*** in the choice sample, and 12.555*, 8.400, and 9.555 in the recoded choice sample (***,**,* denote significance at the 1%, 5% and 10% level, respectively). IIA tests when the status quo alternative is removed cannot be computed.

stone pine forests to urban centers. In addition, stone pine forests cover almost 2.5 more hectares than cork oak forests in the studied area. The attributes *INS*, *ANI* and *MUSH* have positive signs, meaning that respondents are more likely to choose a forest visit alternative with these additional features. *ENTP* and *EXPP* both offer negative signs, indicating that the probability of choosing a recreation visit alternative decreases with higher payments.

Table 4.3. Choice and recoded choice models. Likelihood ratio tests for comparing parameter vectors

Attribute	Choice (CH)			Recoded choice (RC)		
	Conditional logit	Mixed logit		Conditional logit	Mixed logit	
	Mean parameter	Mean parameter	St. dev. parameter	Mean parameter	Mean parameter	St. dev. parameter
ASC-REC	1.7839***	2.2942***		1.3580***	1.9211***	
	(0.3091)	(0.4716)		(0.2897)	(0.4911)	
TREE	-0.1652**	-0.2000*	0.7318**	-0.0285	-0.0799	1.0466***
	(0.0675)	(0.1213)	(0.3083)	(0.0721)	(0.1529)	(0.3480)
INS	0.8737***	1.4375***	1.3474**	0.8108***	1.4771***	1.9569***
	(0.1379)	(0.3446)	(0.5325)	(0.1469)	(0.3973)	(0.5983)
ANI	0.5079***	0.7939***	2.0877***	0.3674**	0.5603**	1.5334***
	(0.1368)	(0.3044)	(0.5850)	(0.1454)	(0.2794)	(0.5307)
MUSH	0.2712**	0.5492*	1.6511***	0.5380***	0.9777***	1.5225***
	(0.1320)	(0.2883)	(0.5731)	(0.1429)	(0.3132)	(0.5304)
ENTP	-0.1402***	-0.2400***		-0.1560***	-0.2858***	
	(0.0222)	(0.0537)		(0.0263)	(0.0644)	
EXPP	-0.0517**	-0.0859***		-0.0695***	-0.1307***	
	(0.0204)	(0.0332)		(0.0232)	(0.0436)	
n	324	324		286	286	
Log-likelihood	-351.67	-339.75		-325.31	-312.89	
Adj. McFadden ρ^2	0.1259	0.1520		0.1076	0.1376	
Likelihood ratio test	H_{A1}: $\beta_{CH}=\beta_{RC}$	Reject H_{A1}?	H_{A2}: $\mu_{CH}=\mu_{RC}$	Reject H_{A2}?	Reject H_A? $\mu\beta_{CH}=\mu\beta_{RC}$	
Conditional logit						
χ^2 (CH vs. RC)	7.520	No	1.042	No	No	
Mixed logit						
χ^2 (CH vs. RC)	9.492	No	2.755	No	No	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.

Standard deviation parameters are all significant, with the highest values found for the attributes *ANI* and *INS*. This indicates more heterogeneity for preferences that lean toward the presence of animals and recreation infrastructures. According to the adjusted McFadden ρ^2, choice models are better adjusted than recoded choice models and the mixed logit outperforms the conditional logit. In most cases, the estimated parameters show no large differences between choice and recoded choice models. The Likelihood Ratio test shows that we cannot reject the hypothesis (H_A) that the parameter vectors of the choice and recoded choice models are statistically indistinguishable for both model specifications (Table 4.3).

The complete combinatorial test indicates that we cannot reject the hypothesis (H_B) of statistically indistinguishable WTP in most cases (Table 4.4). This is rejected only for the attributes *ASC-REC* and *TREE* and the Stone Pine visit in the conditional logit model when using the *ENTP* parameter, and for the attribute *TREE* in the conditional logit model when using the *EXPP* parameter. Overall, our results show convergent validity between choice and recoded choice. In the following analyses we work with a data-enriched model that pools the choice and recoded choice datasets.

A caveat from our results is that the size of our subsamples are relatively small, decreasing the statistical power of the tests and therefore the probability of rejecting the null hypothesis. If we have larger subsamples the results of the tests may change. However, recent studies run similar tests using larger samples sizes[101] and do not reject the null hypothesis of similar results between choice and recoded choice (from a ranking).

[101] Caparrós et al. (2008) use subsamples of 3600 and 3594 observations (450 and 447 respondents, respectively) and Akaichi et al. (2013) use subsamples of 688 and 736 observations (43 and 46 respondents, respectively).

Table 4.4. Mean willingness to pay (WTP) values and complete combinatorial test results from choice and recoded choice models

Attribute	Choice (CH)		Recoded choice (RC)		Complete combinatorial test (H_B: $WTP_{CH,k,pv} = WTP_{RC,k,pv}$)	
	Conditional logit (WTP)	Mixed logit (WTP)	Conditional logit (WTP)	Mixed logit (WTP)	Conditional logit (*p-value*)	Mixed logit (*p-value*)
Entrance fee payment vehicle						
ASC-REC	13.07	9.68	8.96	6.86	0.069*	0.146
	[9.17, 17.98]	[6.68, 13.40]	[5.84, 12.69]	[4.30, 9.92]		
TREE	-1.23	-0.82	-0.20	-0.29	0.075*	0.250
	[-2.20, -0.40]	[-1.76, -0.05]	[-1.03, 0.57]	[-1.28, 0.66]		
INS	6.37	6.09	5.33	5.31	0.277	0.340
	[4.52, 8.75]	[4.05, 8.51]	[3.56, 7.60]	[3.35, 7.70]		
ANI	3.67	3.47	2.37	2.07	0.182	0.149
	[2.10, 5.49]	[1.53, 5.66]	[0.85, 4.10]	[0.45, 3.85]		
MUSH	2.00	2.40	3.57	3.51	0.143	0.254
	[0.42, 3.68]	[0.34, 4.63]	[1.92, 5.48]	[1.85, 5.49]		
Stone Pine visit	14.30	10.50	9.16	7.15	0.069*	0.116
	[10.17, 19.35]	[7.36, 14.34]	[6.02, 13.00]	[4.44, 10.35]		
Cork Oak visit	11.84	8.86	8.76	6.58	0.171	0.205
	[8.13, 16.49]	[5.70, 12.65]	[5.68, 12.55]	[3.85, 9.72]		
Increased trip expenditures payment vehicle						
ASC-REC	39.70	28.72	22.72	16.37	0.142	0.117
	[19.99 , 86.45]	[16.24, 53.73]	[11.87, 40.46]	[9.20, 27.48]		
TREE	-3.76	-2.47	-0.53	-0.68	0.086*	0.217
	[-10.10 , -0.86]	[-6.49, 0.14]	[-2.66, 1.32]	[-3.05, 1.63]		
INS	20.38	18.63	13.80	13.19	0.272	0.270
	[8.72 , 45.33]	[8.23, 37.52]	[6.51, 26.86]	[6.08, 24.58]		
ANI	11.65	10.36	5.95	4.95	0.176	0.149
	[4.73 , 24.03]	[3.67, 21.41]	[1.88, 11.68]	[1.10, 10.46]		
MUSH	6.81	7.02	9.22	8.73	0.292	0.403
	[0.78 , 15.90]	[0.85, 16.88]	[3.87, 17.91]	[3.56, 16.79]		
Stone Pine visit	43.46	31.19	23.26	17.05	0.118	0.102
	[22.03 , 95.86]	[17.35, 58.91]	[12.09, 41.00]	[9.35, 29.46]		
Cork Oak visit	35.94	26.26	22.19	15.69	0.178	0.148
	[18.22 , 77.99]	[13.72 , 49.79]	[11.54 , 40.28]	[8.32, 26.83]		

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets; asterisk (e.g.,*) denote significance at the 10% level; subscript *k* stands for attribute *k*, and subscript *pv* stands for payment vehicle.

4.4.3. Entrance fee versus increased trip expenditures

The pooled models show similar results as the previous models in terms of significance and sign of the parameters (Table 4.5). We also find that the negative parameter is larger for *ENTP* than it is for *EXPP*. This indicates that having to pay a specific amount of money as an entrance fee derives a lower probability of visiting the forest than having to pay the same amount of money as additional trip expenditures. It seems then that respondents considered these two payment vehicles separately in their decision instead of the aggregated payment resulting from adding them up. A *t-test* for differences between *ENTP* and *EXPP* parameters shows significant differences in both the conditional logit (t-stat = –119.28; p-value < 0.0001) and the mixed logit model (t-stat = –102.94; p-value < 0.0001). Thus, we reject the hypothesis (H_C) of similar marginal utility of money associated with these payment vehicles.

Table 4.5. Pooled models using the choice and the recoded choice data

Attributes	Conditional logit	Mixed logit	
	Mean parameter	Mean parameter	St. dev. parameter
ASC-REC	1.5505***	2.0012***	
	(0.2099)	(0.3248)	
TREE	-0.1027**	-0.1574*	0.8351***
	(0.0491)	(0.0937)	(0.2248)
INS	0.8387***	1.4051***	1.6360***
	(0.1000)	(0.2515)	(0.3806)
ANI	0.4357***	0.6450***	1.8748***
	(0.9897)	(0.1978)	(0.3805)
MUSH	0.3964***	0.7461***	1.3031***
	(0.9642)	(0.2025)	(0.3962)
ENTP	-0.1436***	-0.2525***	
	(0.1680)	(0.0394)	
EXPP	-0.0580***	-0.0967***	
	(0.1519)	(0.0254)	
n	610	610	
Log-likelihood	-681.26	-660.14	
Adj. McFadden ρ^2	0.1180	0.1435	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.

The WTP values obtained with each payment vehicle (Table 4.6) show significant differences in all cases, except for the attribute *TREE*. This rejects the hypothesis (H_D) that both payment vehicles derive a similar WTP estimates. Differences imply that respondents are willing to pay around 2.7 additional Euros as trip expenditures for each additional euro they are willing to pay as an entrance fee for visiting the forest. Although our design allows individuals to choose according to the aggregated cost from both payment vehicles, our models show that their utility parameters are significant and different from each other. If respondents had made the choice considering the aggregated costs, the utility parameters of the payment vehicles (and therefore the derived WTP estimates) should have been similar.

Table 4.6. Mean willingness to pay (WTP) values and complete combinatorial test results obtained with the entrance fee and the increased trip expenditures payment vehicles from the pooled model

Attributes	Conditional logit		Mixed logit		Complete Combinatorial test (H_D: $WTP_{ENTP,k} = WTP_{EXPP,k}$)	
	Entrance fee (WTP)	Increased trip expenditures (WTP)	Entrance fee (WTP)	Increased trip expenditures (WTP)	Conditional logit (*p-value*)	Mixed logit (*p-value*)
ASC-REC	10.90	28.46	8.15	21.73	0.001***	0.000***
	[8.35, 13.72]	[18.58, 44.52]	[6.18, 10.33]	[14.62, 33.04]		
TREE	-0.73	-1.97	-0.54	-1.46	0.133***	0.224**
	[-1.31, -0.17]	[-4.14, -0.41]	[-1.24, 0.11]	[-3.45, 0.32]		
INS	5.92	15.68	5.58	15.04	0.002***	0.001***
	[4.56, 7.49]	[9.26, 25.44]	[4.15, 7.10]	[9.33, 24.42]		
ANI	3.07	7.99	2.58	6.87	0.013**	0.038***
	[1.86, 4.34]	[4.38, 13.10]	[1.25, 3.90]	[3.03, 11.74]		
MUSH	2.75	7.22	2.89	7.67	0.021**	0.023**
	[1.64, 4.02]	[3.74, 12.36]	[1.58, 4.34]	[3.89, 12.44]		
Stone Pine visit	11.63	30.43	8.69	23.19	0.000***	0.000***
	[9.04,14.72]	[19.87, 47.21]	[6.60, 11.06]	[15.45, 34.97]		
Cork Oak visit	10.16	26.49	7.60	20.27	0.001***	0.001***
	[7.58, 13.07]	[16.88, 41.24]	[5.56, 9.85]	[13.52, 31.23]		

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets; asterisks (e.g.,***,**) denote significance at the 1% and 5% level, respectively; subscript *k* stands for attribute *k*.

The rejection of convergent validity for these payment vehicles opens up the question of why this occurs. A possible explanation is related to the perception of the *right* to access forest areas freely in Spain. Free access to forests for public users in Spain is possible in some areas owned by public administrations, while the remaining private and public forests are not open to the general public. However, the possibility of accessing some areas freely may have created the perception for some people that they have the right to access forests for free. This may have resulted in protest behavior towards the entrance fee payment vehicle, reducing the stated WTP. Campos et al. (2007) discuss that this is the cause of different protest response rates between different subsamples facing one of these payment vehicles (up to 35% of protest for the entrance fee and 3% of protest for the increased trip expenditures) as most protest responses for the entrance fee were related to the perceived right to freely access forest areas. In our choice experiment, the protest response rate is lower (6%) than in Campos et al. (2007), but the difference between WTP estimates has not been reduced. A possibility is that the same protest behavior found in Campos et al. (2007) is present in our results but translated into an effect on the WTP, which is positive but significantly reduced respect to the increased trip expenditures. Another explanation is that the entrance fee payment vehicle may be more subject to strategic behavior; some respondents may think that by stating a lower payment for the entrance fee they may influence future policies aiming to establish fees for accessing these areas. Unfortunately, we did not design the survey with the goal of identifying these or other factors that drive this discrepancy in WTP estimates. This is left for future research with this specific goal in mind.

Given the results obtained, the size of the subsamples in this analysis is not as relevant as in the comparison of the choice and the recoded choice. Having larger subsamples would increase the statistical power of the payment vehicle tests, but this would make even more likely the rejection of our null hypothesis.

The mean WTP values from Table 4.6 lie within the range of values estimated in recent studies valuing public recreation in Spanish forests and using the same payment vehicles than our choice experiment. Studies using the entrance fee (Arriaza et al., 2002; Campos et al., 2007; Samos Juarez and Bernabeu Cañete, 2011; Martínez-Carrasco et al., 2012) offer mean WTP values between €4 and €12 (updated to 2008 Euros), so our estimations are in the upper bound of this range. The only study using the increased trip expenditures (Campos et al., 2007) offers a mean WTP between €14 and €23 (updated to 2008 Euros), so our estimates are in the upper bound of these values, and in some cases slightly higher. We must note that these previous studies employed the contingent valuation method, and that choice experiments usually offer higher mean WTP values than contingent valuation (Metcalfe et al., 2012; Oviedo and Caparrós, 2015).

4.4.4. Aggregated results

We estimate the annual visits to stone pine forests from a question included in the survey. This question asked respondents who had made a recreation visit to a Spanish forest at least once in the past 12 months how many of these visits they made to stone pine forests. Based on this information, and considering the population from each of the provinces in our stratified sample, we estimate 13,359,885 visits to stone pine forests between May 2007 and April 2008 (we assume that these visits correspond to 2008). As the stone pine forest area in Spain (excluding the northeast area) is 451,826 hectares (MAGRAMA, 2015a)[102], we obtain 29.57 visits per hectare in 2008. For cork oak forests, we use the data on annual visits to the Monfragüe National Park and to the Alcornocales Natural Park, the two major cork oak forest areas that receive recreation visits in the southwest and west of Spain. For the Monfragüe National Park, MAGRAMA (2015b) estimates 331,788 visits in 2008. For the Alcornocales Natural Park, Oviedo et al. (2015) calculated 1,737,695 visits in 2010. In the latter case, we assume that the number of visits were similar in 2008. As these two forests cover an area of 186,163 hectares (Europarc España, 2012), we estimate 11.11 visits per hectare in 2008.

To obtain aggregated values (in 2008 Euros), we work with the pooled dataset. Additionally, instead of using the payment vehicles *ENTP* and *EXPP* separately, this model includes the variable *Bid* that adds up the amount offered with both payment vehicles. This is motivated by the fact that, due to the observed divergences between payment vehicle results, we are unable to pick a single payment vehicle to obtain aggregated values. As Campos et al. (2007) discusses, both vehicles can be conditioned by effects associated with the payment context that they create. We believe that adding up both payments in the model is the best solution for estimating aggregated values. The results of this model show that all parameters are significant and have the same signs as in previous models (Table 4.7). The mixed logit is better adjusted than the conditional

[102] This only considers hectares where the stone pine is identified as the main species.

logit according to the McFadden ρ^2. The significance of the standard deviation parameters for *TREE, INS, ANI* and *MUSH* also shows preference heterogeneity for these attributes.

Table 4.7. Pooled models using the choice and recoded choice data and including the variable *Bid* as payment attribute

Attribute	Conditional logit	Mixed logit	
	Mean parameter	Mean parameter	St. dev. parameter
ASC-REC	1.5446***	1.9192***	
	(0.2087)	(0.2999)	
TREE	-0.0993**	-0.1461*	0.7528***
	(0.0488)	(0.0869)	(0.1993)
INS	0.7918***	1.2452***	1.5240***
	(0.0985)	(0.2153)	(0.3505)
ANI	0.4326***	0.6189***	1.6524***
	(0.0981)	(0.1839)	(0.3462)
MUSH	0.3982***	0.6896***	1.2751***
	(0.0958)	(0.1865)	(0.3594)
Bid	-0.0978***	-0.1586***	
	(0.0112)	(0.0228)	
n	610	610	
Log-likelihood	-688.69	-668.41	
Adj. McFadden ρ^2	0.1089	0.13321	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively; the attribute *Bid* adds up the two amounts offered in the entrance fee and increased trip expenditures payment attributes.

In all cases the conditional logit model offers a higher mean WTP for the *CV* calculation, a higher p^* for the *SEV* calculation and higher aggregated values than the mixed logit model (Table 4.8)[103]. For the mixed logit we find a higher mean WTP for the *CV* calculation and a higher p^* for the *SEV-2* calculation for stone pine forests. The percentage of visits in *SEV-2* is lower in stone pine forests (45% versus 46% for cork oak forests). Aggregated values are higher in all cases for stone pine forests because the number of annual visits to these forests is six times higher than the number of visits to cork oak forests. For the *SEV-3* scenario, p^* is forced to be the same for

[103] Note that we round to the nearest unit without decimals the estimated *p** values in Table 4.8, as we expect that the price in the potential market would be also rounded.

both forests, but it translates to a different percentage of visits in each forest. Compared to *SEV-2*, this percentage drops to 33% for stone pine forests and to 27% for cork oak forests. Aggregated values are higher for stone pine forests in SEV-3 in all cases.

If we compare the three aggregated measures, we see that the CV offers the higher value, the *SEV-2* represents between 45 and 51% of the *CV*, and the *SEV-3* represents between 68 and 90% of the *SEV-2*. As expected, both the market simulation and the inclusion of substitutes imply a lower aggregated value for forest public recreation. Overall, the simulated exchange value is between 35 and 51% of the compensating variation value.

Finally, we also notice that the survey was conducted in 2008 and the estimated economic values may have varied significantly since then. While using the consumer price index variation over a period may be appropriate to update values, it is true that the economic recession and the slow economic recovery since 2008 may have changed social preferences for non-market services. Therefore, our estimations must be taken carefully. However, we think that this time difference between survey and publication dates is not particularly relevant for the validity test and for the comparison of economic measures presented, as any variation in values should not be affecting differently to the compared estimates.

Table 4.8. Aggregated values of compensating variation (*CV*) and simulated exchange value (*SEV*) of public recreation in stone pine and cork oak forests in Spain (year 2008)

Measure	Stone pine				Cork oak			
	€ per visit	Visits	Aggregated		€ per visit	Visits	Aggregated	
			€	€/ha			€	€/ha
Conditional logit								
CV	16.81	13,359,885	224,564,914	497.02	14.78	2,069,483	30,583,277	164.28
SEV-2[a]	19.00	5,966,946	113,371,981	250.92	18.00	873,059	15,715,055	84.42
SEV-3[b]	22.00	3,837,585	84,426,859	186.86	22.00	487,378	10,722,323	57.60
Mixed logit								
CV	13.02	13,359,885	173,939,696	384.97	11.18	2,069,483	23,141,404	124.31
SEV-2[a]	13.00	6,011,948	78,155,327	172.98	12.00	951,962	11,423,546	61.36
SEV-3[b]	16.00	4,408,762	70,540,193	156.12	16.00	558,760	8,940,167	48.02

Note: Total hectares of stone pine and cork oak forests are 451,826 and 186,163, respectively.
[a] *SEV-2* stands for simulated exchange value in the scenario of a forest visit alternative and the status quo.
[b] *SEV-3* stands for simulated exchange value in the scenario of two forest visit alternatives (either to a stone pine or to a cork oak forest) and the status quo.

4.5. Intermediate summary

We have presented two convergent validity tests of a choice experiment for valuing public recreation in stone pine and cork oak forests in Spain. On the one hand, we have obtained similar results from a choice and from a ranking recoded as a choice in an exercise involving three alternatives plus the status quo. On the other hand, we have found that respondents are willing to pay €2.7 more as increased trip expenditures for each additional euro paid as an entrance fee. While these additional payments would have the same effect on the disposable income of forest recreationists, they do not seem to have the same effect on their utility. These results corroborate the findings from Campos et al. (2007) who used the contingent valuation method. These differences may be explained by the perception of some forest recreationists that they have the right to freely access the forest and by the strategic behavior of some respondents who try to avoid the establishment of fees for accessing current free access areas by stating a lower WTP when facing the entrance fee payment vehicle.

Our empirical results show a small but significant difference in the WTP per person for visiting stone pine and cork oak forests in Spain. The WTP per person is around €2 higher in stone pine forests. The valuation of the other attributes shows a significant and positive WTP for the possibility of seeing domesticated animals, the possibility of picking mushrooms, and for infrastructures that foster recreation opportunities. Of these three attributes, the highest value is placed on infrastructures. An implication for managers is that a higher priority should be given to forest areas with presence of these types of infrastructures.

Concerning the aggregated economic values, we compare the compensating variation and the simulated exchange value measures. The latter measures account for 35 to 51% of the former values. Aggregated values are higher for stone pine forests as they receive more visitors than cork oak forests. Results indicate that when visitors face a scenario in which they must actually pay for enjoying forest recreational activities, not all of them would be willing to pay this price. In particular, when there is no any other forest alternative, only around 44.82% of visitors of stone pine forests and 44.09% of visitors of cork oak forests would pay the price. When a forest visit alternative exists, percentages drop to 30.86% and 25.27% of visitors of stone pine and cork oak forests, respectively.

CHAPTER 5. AN APPLICATION TO A CONSERVATION PROGRAM FOR THE IBERIAN LYNX IN SPAIN

5. AN APPLICATION TO THE IBERIAN LYNX CONSERVATION IN SPAIN[104]

The integration of non-market ecosystem services into the national accounts in a consistent way with market goods and services has shown to be possible for public forest recreation by using the SEV method, as discussed in the previous chapter. The main goal of this chapter is to apply this methodology to the economic valuation of threatened biodiversity, which is also a public good. In particular, we are now interested in analyzing how Spanish people assess a conservation program to ensure that the Iberian Lynx (*Lynx pardinus*) returns to 1990 levels, as well as the role they adopt when facing a contribution to an environmental program.

As explained throughout the dissertation, the SEV method requires the estimation of the demand and supply functions in order to calculate a (*simulated*) exchange value. In this chapter, we use a contingent valuation study to obtain the demand function for this conservation program for the Iberian Lynx. Contingent valuation has been widely used in assessing individuals' WTP for environmental public goods such as the one we are dealing with in this application.

Next to this, motivations for paying for environmental public goods may differ among agents. If individuals use different response patterns, they are potentially using multiple preference orderings that provide different answers depending on the context the valuation is framed (Arrow, 1951; Harsanyi, 1955; Sen, 1977; Margolis, 1982; Hausman and McPherson, 1996; Beckerman and Pasek, 1997; Nyborg, 2000; Russell et al., 2003; Eggert et al., 2016). When the good or service in question solely affects the own respondent's well-being, it is likely that he/she is acting as consumer; while when it comes to pure public

[104] This chapter is based on Ruiz-Gauna et al. (2016).

goods, he/she may be acting either as a consumer or as a citizen (Sagoff, 1988; Blamey et al., 1995; Nyborg, 2000).

Altruistic behavior related to the consumer/citizen dichotomy has been widely studied both from a general point of view (Edwards, 1986; Andreoni, 1990; Holmes, 1990; Johansson, 1993, Arrow et al., 1993; Crowards, 1997; Curtis and McConnell, 2002) and with regard to environmental conservation and wildlife preservation (Sagoff, 1988; Stevens et al., 1991, 1993; Blamey et al., 1995). As far as we know, the only study formalizing the differentiation between consumers and citizens can be found in Nyborg (2000). She suggests using the term *Homo Economicus* (henceforth referred to as HE) to describe those individuals who maximize their own wellbeing, and the term *Homo Politicus* (henceforth referred to as HP) to describe those individuals who maximize social welfare and respond as they think that society should act. The latter individuals are motivated by altruism and consider that the entire society has a moral obligation of conserving the environment. Such a distinction suggests that every individual may have two different and possibly conflicting preference orderings depending upon whether they behave as HE or as HP. For example, one individual may prefer one alternative from a social perspective, while another may prefer the same alternative from a personal point of view. Nyborg (2000) goes a step further and also differentiates two types of individuals within *Homo Politicus*. She terms *Homo Politicus with shared responsibility* those who consider that the whole society has a moral obligation for providing the good (the decision of other citizens does matter), and *Homo Politicus with sole responsibility* those who act independently from the rest (personal consideration concerning environmental goods). These two types of individuals are henceforth referred to as HP_{sh} and HP_{so}, respectively.

In our study, we introduce another type of individual: *Homo Strategicus* (or free-rider), henceforth referred to as FR. While these individuals may be interested in environmental conservation, they expect the payment of others may be enough for achieving the goal. In other words, they act knowing the possibility of getting some advantage with the parts provided by other respondents, thereby having a strategic incentive to understate WTP (McFadden and Leonard, 1993). This kind of behavior is considered to be rational in one shot games in a non-cooperative game theory framework. This is relevant because, contrary to what may happen with market goods and services, when it comes to public goods, it does not seem so obvious that social welfare will be promoted following only self-interest because many individuals (maybe all of them) would want to free-ride. While theory predicts free-riding, empirical research provides mixed results (Marwell and Ames, 1981; Kim and Walker, 1984; Andreoni, 1988, 1995; Weimann, 1994; Ledyard, 1995; Bardsley, 2000; Champ et al., 2002; Fischbacher and Gächter, 2010).

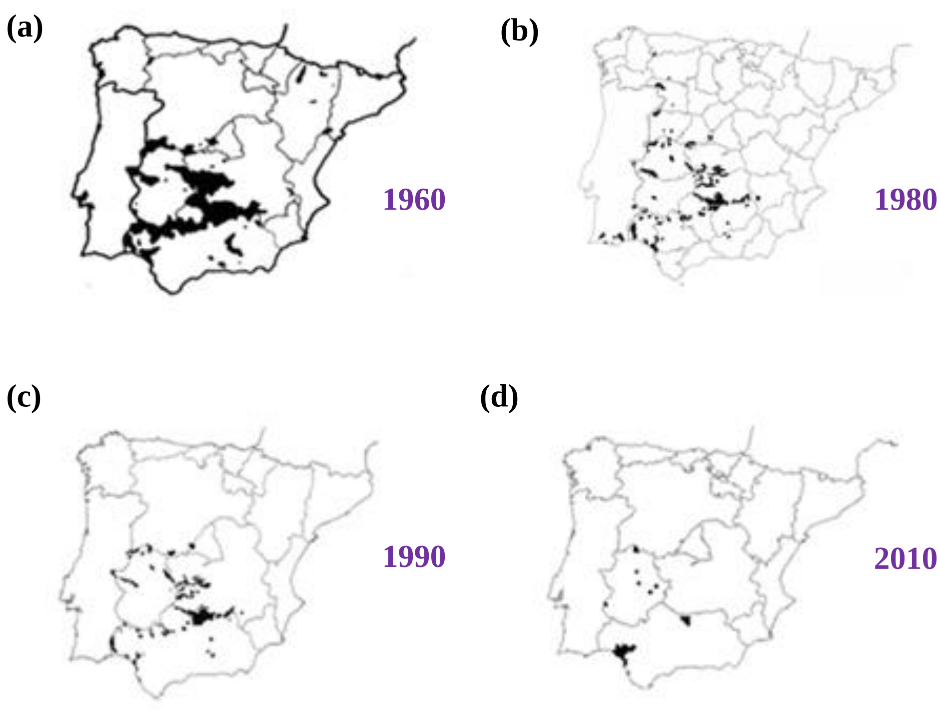
After empirically identifying this triple consumer-citizen-strategic distinction through follow-up questions in the contingent valuation survey, we look at how it affects the valuation function of the respondents' WTP for the Iberian Lynx conservation program and the aggregation of the estimated values. While applications of contingent valuation to biodiversity conservation usually focus on the compensating variation (or any other Hicksian variation), we extend this analysis by also applying the SEV method.

5.1. Survey and design

The Iberian Lynx is listed as one of the most endangered feline species of the planet by the International Union for Conservation of Nature (IUCN). In fact, it was classified as critically endangered in 2008 according to the IUCN Red List criteria; while it became classified as endangered in 2015 (Rodríguez and Calzada, 2015).

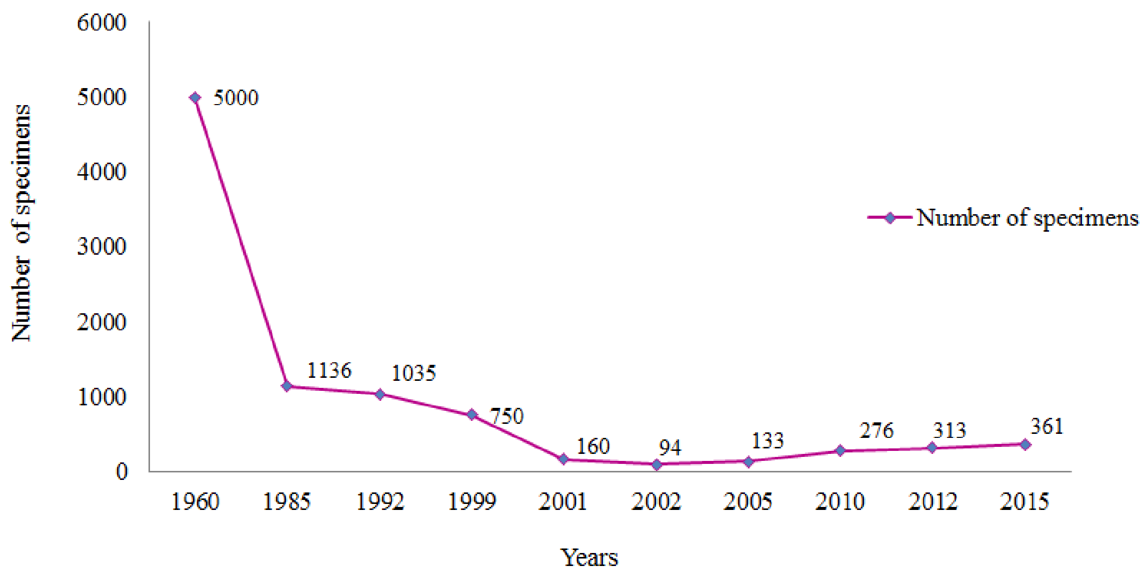
Although this species is confined to the Iberian Peninsula, from where it originated, it is currently restricted to central and southwestern Spain and, in particular, to two separate regions, namely Andújar-Cardeña and Doñana-Aljarafe (Simón et al., 2012) (see Figure 5.1). Population fragmentation and local extinction processes have been occurring since the middle of the last century. The population was estimated at 1,100 specimens in 1990, but wild populations have undergone a substantial decline by around 90% in last 20 years (see Figure 5.2). That is why the interest in its conservation has been continually increasing, and several programs, actions and plans for its recovery have been conducted by different administrations and NGOs in recent years (see, e.g. the projects Life + IBERLINCE and Programa de Conservación Ex-Situ del Lince Ibérico) [105].

[105] See, Ferreras et al. (2010), Palomares et al. (2011) and Simón et al. (2012) for a review of different actions.



Source: Rodriguez and Delibes (1990) for figures (a) and (c) and own elaboration based on MAGRAMA (2016) for figure (b) and (d)

Figure 5.1. Spatial and temporal evolution of Iberian Lynx specimens (black areas indicate the Iberian Lynx distribution)



Source: own elaboration based on Rodríguez and Delibes (1990); Guzmán et al. (2005); Simón (2016)

Figure 5.2. Evolution of Iberian Lynx specimens

Our field study deals with a conservation program that seeks for getting back the Iberian Lynx population to 1990 levels, which is considered the minimum threshold necessary to ensure its preservation. The survey was conducted with Spanish adults (> 18 years old) from 14 provinces (*Cádiz, Málaga, Sevilla, Córdoba, Huelva, Badajoz, Cáceres, Valladolid, Madrid, Segovia, Toledo, Salamanca, Zamora and Ávila*) close to areas comprising the potential habitat for the Iberian Lynx or in which sometimes the Iberian Lynx used to exist (see Figure 5.1.b). The Iberian Lynx is very selective when settling in a certain area. This species preferably selects Mediterranean forests and scrublands, that is, open grasslands with a high density of shrubs and trees such as cork oak (*Quercus suber*) and holm oak (*Quercus ilex*). It is usually located 400 to 900 meters above the sea level, where there is a low and dense scrub brush, thus avoiding farmlands and population centers. This habitat provides the adequate shelter and access to open pasture for the Iberian Lynx to feed on the European Rabbit (*Oryctolagus cuniculus*), which represents 90% of their diet (Palomares et al., 2000; Palomares, 2001).

A professional surveying company interviewed 750 individuals at their homes from May to July 2008. A booklet (see Appendix G) was presented to the respondents, with a brief description of the Iberian Lynx. The main highlights were: a) it is the most endangered feline species along with the Bengali tiger in the world; b) population has decreased from 1,100 to 200 individuals over the last 20 years; and c) 1,100 individuals are considered as the minimum threshold for the preservation of this species. The survey response rate was 70%. We also identify 93 protest responses (12.4% of the sample) giving a valid sample size of 657 individuals. However, our final sample is made up of 596 respondents, as we only consider those respondents who are willing to pay certain amount of money, as we will see below. We also made a pre-test with 50 individuals.

The valuation question was asked in a double-bounded format in which the first question asked respondents whether they would be willing to pay a certain amount of money to implement a program to conserve the Iberian Lynx (see Question 15 in Appendix H). If the answer was 'yes', they were offered the possibility of paying a larger amount (Question 16 in Appendix H). If the answer was 'no', they were offered the possibility of paying a lower amount (Question 17 in Appendix H). Following-up, we included an open-ended question that asked respondents to indicate the maximum amount of money they would be willing to pay (Question 18 in Appendix H). We have only used this question to identify the number of individuals who are willing to pay.

The bid offered in the double-bounded question varies from one survey to another. We used three different bid vectors: (10; 20; 30; 45) for the single-bounded question, (30; 35; 45; 65) for the upper bound of the double-bounded question, and (5; 10; 15; 20) for the lower bound of the double-bounded question (see the original survey in Appendix B).

5.1.1. Payment vehicles

Payment vehicles are usually compared in split-sample designs of contingent valuation so that a question including a different payment vehicle is randomly assigned to each respondent. Previous valuation research on the effects of payment vehicles in contingent valuation finds evidence of statistical significant differences between estimations (Johansson, 1996; Morrison et al., 2000; Jakobsson and Dragun, 2001; Champ et al., 2002; Bergstrom et al., 2004; Campos et al., 2007; Wiser, 2007; Ivehammar, 2009; Aoun, 2015). Kontoleon et al. (2005) is an exception to this. Most of these papers use taxes and donations (or voluntary contributions) as payment mechanisms. Champ et al. (2002) also incorporate the possibility of holding a referendum and Kontoleon et al. (2005) compare a standard production tax with a tax reallocation scheme. However, only a few of them are directly focused on the valuation of biodiversity conservation-related issues (Johansson, 1996; Jakobsson and Dragun, 2001; Aoun, 2015).

In our exercise, we compare the results from four payment vehicles: 1) contribution to a fund managed by a NGO; 2) contribution to a fund managed by a NGO with a guarantee that funds would be reimbursed if not enough were collected (if more than enough were collected, the surplus would be repaid); 3) additional increased income tax; and 4) a referendum for an increased income tax (the project will only be launched if a majority votes for it; see Appendix H). The payment associated to each of these four payment vehicles would be implemented only in one year. As the first two and the second two payment vehicles are similar to each other, we also analyze the same scenario by grouping them into two categories: contribution to a fund and increased income tax. Table 5.1 shows the number of individuals facing each payment vehicle in our sample.

Table 5.1. Number of individuals facing each payment vehicle

Four payment vehicles	Contribution to a fund	Contribution to a guaranteed fund	Increased income tax	Referendum for an increased income tax
	198	178	182	192
Two payment vehicles	Contribution to a fund		Increased income tax	
	376		374	

5.1.2. Motivations for paying to conserve the Iberian Lynx

Motivations for paying may be very different when individual preferences come into play. To analyze this, we first need to identify how many individuals are actually willing to pay a certain amount of money by building on the single-bounded, double-bounded and open-ended questions mentioned above. According to this format, individuals are willing to pay if they give: (i) a 'yes' response to the single-bounded question, or (ii) a 'no' response to the single-bounded question, but a 'yes' response to the double-bounded question, or, (iii) a 'no' response to the single-bounded question, a 'no' response to the double-bounded question, but a positive (non-zero) amount of money to the open-ended question.

By contrast, they would not be willing to pay if they give a 'no' response to the single-bounded question, a 'no' response to the double-bounded question and either a zero response or a 'no' response to the open-ended question.

Based on this information, we can build different models in order to identify the behavior of individuals when facing a payment scenario to protect the Iberian Lynx. To that end, we will use a set of follow-up questions.

<u>MODEL HEPFR</u>: In this scenario we differentiate between HE and HP. Note that only those responding paying a certain amount of money can be classified as HE or as HP.

In order to know whether a respondent behaves as a consumer or as a citizen, we use a HEPFR follow-up question (see Question 19 in Appendix H). It has two possible answers:

a) I pay because I consider socially correct to guarantee the preservation of the Iberian Lynx.
b) I pay because the preservation of the Iberian Lynx is important for me.

If the response is a), respondent is classified as HP; whereas if the response is b), respondent is classified as HE.

However, it should be pointed out that this model includes the possibility of a free-rider problem, as there may be individuals considered as HE or HP who behave strategically. To the extent that we work with the whole sample in this model, we do not analyze this effect separately. Determining how many of these respondents act as free-riders and excluding them from the sample allows us to construct the following model:

<u>MODEL HEP</u>: This scenario differentiates between HE and HP, while eliminating the free-riding effect. That is, it only considers those HE and HP who do not behave strategically.

In other words, HE and HP may be divided into those who are not FR and those who are FR. Thus, if we want to determine how many of them do not adopt this strategic behavior, we need to look at the HEP follow-up question (see Question 20 in Appendix H). This question has four possible answers:

a) I thought that if I paid the rest of the Spanish citizens (contributors) would pay the same quantity.
b) I thought that if I paid the rest of the Spanish citizens (contributors) would pay, but not necessarily the same quantity.
c) I thought that if I paid even a small quantity, this could be compensated by the payment of other contributors.
d) I did not take into account the quantity that others would pay.

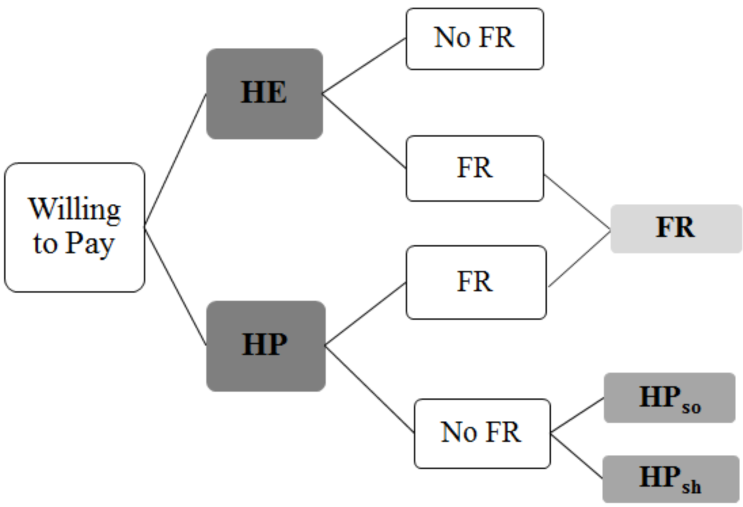
If the response is c), respondent is identified as a free-rider.

But, what happens with respondents answering a), b) or d)? Such responses help us build the third model:

MODEL HSSFR: In this scenario, we differentiate between HE, HP_{sh}, HP_{so} and FR.

Coming back to the HEP follow-up question, if response is d), respondent is HP_{so}; whereas if the response is a) or b), respondent is HP_{sh}.

In short, the five types individuals analyzed in our exercise are shown in Figure 5.3 (in grey color). This figure also permits visualizing graphically the three models referred to.



Source: own elaboration

Figure 5.3. Classification of types of individuals

The next issue consists in analyzing what happens with those respondents who are not willing to pay any amount of money. Again, it may also be the case that some respondents are considered as FR even if they are not willing to pay. In order to identify them, we need the FR follow-up question posed to these respondents[106]. This question[107] has three possible answers:

[106] Follow-up questions number 21, 22, 23 and 25 in Appendix H have not been analyzed, as we believe that they may lead to confusion and do not provide more explanatory capacity to the models.

[107] In the case of the type of survey in which the payment vehicle is a referendum for an increased income tax, options are: a) I thought that although I did not vote in favor the rest of the Spanish people would vote in favor, b) I thought that if I did not vote the rest of the Spanish people would not vote either, and c) I did not consider what others would vote.

a) I thought that if I did not pay the rest of the Spanish citizens (contributors) would pay.
b) I thought that if I did not pay the rest of the Spanish citizens (contributors) would not pay either.
c) I did not take into account the quantity that others would pay.

If the response is a), respondent is classified as FR[108].

According to this survey, the number of individuals who belong to each group is shown in Table 5.2.

Table 5.2. Classification of types of individuals by groups

	Observations
Paying	596
Homo Economicus (HE)	279
Strategicus (FR)	32
Homo Politicus (HP)	317
With sole responsibility (HP_{so})	155
With shared responsibility (HP_{sh})	120
Strategicus (FR)	42
Not paying	154
They definitively do not pay	61
Strategicus (FR)	4
Protest responses	93
TOTAL	750

As it can be seen in Table 5.2., there are 93 protest responses (12.4% of the sample). We identified them by asking a follow-up question to respondents who answered 'no' to the single-bounded question, 'no' to the double-bounded question and a zero WTP or 'do not know' answer to the open-ended question. In the follow-up, we asked them the motives of their responses and based on these motives we identified them as 'zero' protests.

[108] In Appendix I we analyze the dichotomy between FR and those who do not follow a strategic behavior, regardless of whether they are willing to pay or not.

5.1.3. Clarity, credibility and certainty of the scenario

The survey also included three questions to identify whether respondents found the contingent valuation scenario difficult (Question 26 in Appendix H), whether they found the possibility of implementing the program through any of the four aforementioned payment vehicles feasible (Question 27) and whether they answered confidently to the contingent valuation question (Question 28). In these questions they were presented with a Likert scale from 1 to 5. For the clarity question, value 1 represents that the question is very unclear while value 5 means the opposite. For the credibility question, value 1 indicates that respondents find impossible that the program be implemented and value 5 means that the start-up of the program is perceived as very possible. For the certainty question, respondents who selected value 1 highly doubted when answering the question, while respondents who assigned a value of 5 answered absolutely convinced.

5.2. Analysis

We now present the econometric specification of the binary discrete-choice format used in our contingent valuation exercise, the comparison tests used and the formulas needed for estimating aggregated economic values for the conservation program for the Iberian Lynx.

5.2.1. Econometric models

We assume a linear-in-parameters utility function for individual *l* and alternative *j* (the preservation of the Iberian Lynx) in a set of *J* alternatives $(j = 1, 2)$[109] with a systematic (V_{lj}) and a random component (ε_{lj}):

$$U_{lj} = V_{lj} + \varepsilon_{lj} = \beta' X_{lj} + \varepsilon_{lj} \qquad [5.1]$$

where β represents a vector of parameters, X_{lj} is a vector of observed variables for alternative *j* and individual *l*; and ε_{lj} are random errors. Depending on the model developed in each case, we include as explanatory variables the variables from Table 5.3 at the end of

[109] Unlike in the fourth chapter, here we only include two values for alternative *j* (paying a certain amount of money to implement the program to protect the Iberian Lynx or not paying).

the section (five types of individuals, four payment vehicles - or two if they are grouped - and three questions related to the clarity, credibility and certainty of the contingent valuation scenario) plus an *Intercept*. Unlike in the fourth chapter, here there is one monetary attribute (*Bid*).

Given the previously defined utility function, the probability that the respondent *l* chooses alternative *j* over any alternative *y* ($\forall y \in J$) ($\Pr_{lj}$) is:

$$\Pr_{lj} = \Pr\left[V_{lj} + \varepsilon_{lj} > V_{ly} + \varepsilon_{ly}\right] = \Pr\left[V_{lj} - V_{ly} > \varepsilon_{ly} - \varepsilon_{lj}\right] \forall j, y \in J \quad [5.2]$$

Using the proposal by Hanemann (1984, 1991) and assuming that $\varepsilon_l = \varepsilon_{ly} - \varepsilon_{lj}$ is logistically distributed, the probability that individual *l* will give a 'yes' answer is:

$$\Pr_{lj} = \frac{1}{1 + e^{-\mu\left(V_{lj} - V_{ly}\right)}} \quad [5.3]$$

which represents a binary logit model. Equation [5.3] is equivalent to saying that:

$$\Pr_{lj} = \frac{e^z}{1 + e^z} = \frac{1}{1 + e^{-z}} = e^z + 1 \quad [5.4]$$

where z is given by the following linear combination $z = \beta_0 + \beta_1 X_1 + ... + \beta_i X_i$, being β_0 the constant coefficient.

The goodness-of-fit of the model is estimated using the maximum log-likelihood ratio. The log-likelihood function of any binary choice model is:

$$\ln L(\beta) = \sum_{l=1}^{n} \left\{ d_{lj}^{yes} \ln \Pr_{lj}^{yes}\left(Bid_l\right) + d_{lj}^{no} \ln \Pr_{lj}^{no}\left(Bid_l\right) \right\} \quad [5.5]$$

where d_{lj}^{yes} is 1 if the response is 'yes' to the payment of an amount of money to accomplish the conservation program and 0 otherwise; while d_{lj}^{no} is 1 if the response is 'no' to any payment and 0 otherwise. As $d_{lj}^{yes} + d_{lj}^{no} = 1$, [5.5] can be rewritten as:

$$\ln L(\beta) = \sum_{l=1}^{n} \left\{ d_{lj}^{yes} \ln \frac{1}{1+e^{-z}} + \left(1 - d_{lj}^{yes}\right) \ln \frac{e^{-z}}{1+e^{z}} \right\} \quad [5.6]$$

Unlike in the single-bounded modeling shown above, in the double-bounded model respondent is presented with two bids. The level of the second bid depends on the response to the first bid. Thus, there are four possible results: both responses are ‘yes’, both responses are ‘no’, a ‘yes’ followed by a ‘no’ and a ‘no’ followed by a ‘yes’. Under the assumption of a utility-maximizing respondent (Hanemann, 1991), the log-likelihood function is:

$$\ln L(\beta) = \sum_{l=1}^{n} \left\{ \begin{array}{l} d_{lj}^{yes,yes} \ln \Pr_{lj}^{yes,yes}\left(Bid_l, Bid_l^{u}\right) + d_{lj}^{no,no} \ln \Pr_{lj}^{no,no}\left(Bid_l, Bid_l^{d}\right) \\ + d_{lj}^{yes,no} \ln \Pr_{lj}^{yes,no}\left(Bid_l, Bid_l^{u}\right) + d_{lj}^{no,yes} \ln \Pr_{lj}^{no,yes}\left(Bid_l, Bid_l^{d}\right) \end{array} \right\} \quad [5.7]$$

where $d_{lj}^{yes,yes}$, $d_{lj}^{no,no}$, $d_{lj}^{yes,no}$ and $d_{lj}^{no,yes}$ are binary-valued indicator variables, Bid_l^u is the upper *bid* and Bid_l^d is the lower *bid*.

We used NLOGIT version 4.0 for estimating the parameters through maximum likelihood for the single-bounded and double-bounded logit functions.

The explanatory variables used in our models are shown in Table 5.3.

Table 5.3. Explanatory variables used to test the different effects in the regression models

Four payment vehicles	
Fund1[a,b]	Takes value 1 if the payment vehicle is contribution to a fund
Fund2[a,c]	Takes value 1 if the payment vehicle is contribution to a guarantee fund
Tax1[a,c]	Takes value 1 if the payment vehicle is increased income tax
Tax2[a,b]	Takes value 1 if the payment vehicle is referendum for an increased income
Two payment vehicles	
Fund [a]	Takes value 1 if the payment vehicle is contribution to a fund
Tax [a]	Takes value 1 if the payment vehicle is increased income tax
Models HEPFR and HEP	
Homo	Takes value 1 if respondent is *Homo Economicus* and 0 if respondent is *Homo Politicus*
Model HSSFR	
HP_{so} [a]	Takes value 1 if respondent is *Homo Politicus with sole responsibility*
HP_{sh} [a]	Takes value 1 if respondent is *Homo Politicus with shared responsibility*
FR [a]	Takes value 1 if respondent is *Homo Strategicus* – both *Homo Economicus* and *Homo Politicus*
Models considering the clarity, credibility and certainty of the contingent valuation scenario	
Clarity[a]	Takes value 1 if respondent answered 3,4 or 5 to the clarity question
Credibility[a]	Takes value 1 if respondent answered 3,4 or 5 to the credibility question
Certainty [a]	Takes value 1 if respondent answered 3,4 or 5 to the certainty question

(a) Takes value 0 otherwise.

(b) The level of the single-bounded and double-bounded questions are €10, €30 and €5, respectively.

(c) The level of the single-bounded and double-bounded questions are €30, €45 and €15, respectively.

HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect. HSSFR refers to the model that differentiates between *Homo Economicus, Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

From these models we also calculate empirical distributions for the individual parameters of each payment vehicle and each type of respondent. For this, we apply the Krinsky and Robb (1986) bootstrapping technique. We take 1,000 random draws from the distribution of the parameters and their variance-covariance matrix. This estimation is then utilized to estimate mean marginal WTP for each payment vehicle and each type of individual by using the following formulas:

$$WTP_{Fund1} = \frac{\beta_{Intercept}}{\beta_{Bid}} + \frac{\beta_{Fund1}}{\beta_{Bid}} X_{Fund1} \quad [5.8]$$

$$WTP_{HE} = \frac{\beta_{Intercept}}{\beta_{Bid}} + \frac{\beta_{Homo}}{\beta_{Bid}} X_{Homo} \quad [5.9]$$

The same procedure is followed for the rest of payment vehicles and for any type of individual. The percentile approach (Efron and Tibshirani, 1993) is applied to estimate the standard deviation and the 95% confidence interval of the estimates.

5.2.2. Comparison tests

Based on the above-mentioned Krinsky and Robb (1986) bootstrapping technique, and using the complete combinatorial test (Poe et al., 2005), we compare the mean marginal WTP measures obtained with each payment vehicle and each type of respondent. We test the following null hypothesis for the payment vehicles:

$$H_E : WTP_{Fund1} = WTP_{Fund2} \; ; \; H_F : WTP_{Fund1} = WTP_{Tax1} \; ; \; H_G : WTP_{Fund1} = WTP_{Tax2}$$

$$H_H : WTP_{Fund2} = WTP_{Tax1} \; ; \; H_I : WTP_{Fund2} = WTP_{Tax2} \; ; \; H_J : WTP_{Tax1} = WTP_{Tax2}$$

$$H_K : WTP_{Fund} = WTP_{Tax}$$

and for the types of individuals

$$H_L : WTP_{HE(HEPFR)} = WTP_{HP(HEPFR)}$$ [For the model HEPFR]

$$H_M : WTP_{HE(HEP)} = WTP_{HP(HEP)}$$ [For the model HEP]

$$H_N : WTP_{HP_{sh}} = WTP_{FR} \; ; \; H_P : WTP_{HP_{so}} = WTP_{FR}$$

$$H_Q : WTP_{HP_{so}} = WTP_{HP_{sh}}$$

We test the same hypothesis for the models that include *Clarity, Credibility* and *Certainty* as explanatory variables (see Table 5.3).

5.2.3. Estimation of aggregated values

As explained in the fourth chapter, the compensating variation (*CV*) is a welfare measure that is used in CBA. The *CV* for each type of individual (Small and Rosen, 1981) is:

$$CV_j = \frac{1}{\beta_{Bid}}\left[\left(\beta' x_0\right) - \left(\beta' x_j\right)\right] \qquad [5.10]$$

where $\left(\beta' x_0\right)$ and $\left(\beta' x_j\right)$ are the part of utility corresponding to the alternative *0* (the status quo) and *j* (the preservation of the Iberian Lynx). In other words, $\left(\beta' x_j\right)$ represents the utility that this alternative provides to the different types of individuals. This allows us to obtain the value of this alternative (note that the type of individual is included as explanatory variable). β_{Bid} is the parameter of the payment vehicle.

We also present the *SEV* measure. As mentioned throughout the dissertation, it assumes that a single price is set for the contribution to the program, which means that only part of the population would pay that price if it were internalized in the market. To that end, this method uses a demand function estimated with non-market valuation techniques (contingent valuation in our application) and a supply function based on the commercial costs associated with the implementation of the program. As in the fourth chapter, we also assume that all costs are fixed as we do not have enough information to simulate the cost function. In this way the maximization of benefits occurs at the same price as the maximization of revenues.

In our hypothetical market, revenues from the program to preserve the Iberian Lynx (R_j) are calculated as the price set for the conservation program (p_j) multiplied by the quantity of contributors[110] (q_j), which depends on the price p_j such that:

$$R_j = p_j \cdot q_j\left(p_j\right) \qquad [5.11]$$

[110] As will be explained below, we consider the population from each of the 14 provinces in our sample.

As in the fourth chapter, we also express the quantity of contributors as the probability of paying a price for taking part in this program (Pr_j) (see equation [4.15]).

The calculation of the estimates under the SEV method is equivalent, from a formal point of view, to that explained in the previous chapter for the scenario of one alternative plus the status quo (*SEV-2*). Hence, we find p_j^* (see Appendix I) when

$$e^{(\beta' x_j)} + 1 + (\beta_{Bid} \cdot p_j) = 0 \qquad [5.12]$$

As this equation has no analytical solution, we have to obtain p_j* by iteration.

5.3. Results

5.3.1. Socioeconomic characteristics of the sample

The average profile of the respondents in our sample is a middle-age married person with an average net family income of almost €1,900, a family of three members, while not even two of these members bring in income to the family. He/she does not hold a college degree and is a salaried employee. He/she does not work in a forestry-related sector. In terms of commitment, most of respondents are deeply gratified to know that the Iberian Lynx will be protected and feel good when giving money on a donation campaign (Table 5.4).

There is no official data available that offer information on the characteristics of these individuals with the Iberian Lynx preservation in Spain. Yet we can contrast our sample characteristics with those from samples of previous studies on the valuation of biodiversity conservation in Spain (Martín-López et al., 2007a; Castro et al., 2011; Álvarez-Farizo et al., 2016) or a specific list of species (Martín-López et al., 2007b), including the Iberian Lynx. In these studies most of respondents are men, with age between 30 and 44 years, net family income between €1,200 and €1,994 (updated to 2008 Euros) and a medium education level (high school) – only in Martín-López et al. (2007a) most of respondents hold a high education level (degree) –. Number of family members in Martín-López et al. (2007b) is 2.91. In Álvarez-Farizo et al. (2016), most of respondents are married and salaried employees. No study offered information about whether they work in the forestry sector. However, this comparison must be taken carefully, as the population sample of these studies is randomly selected from local residents in the Andalusia Region (Castro et al., 2011; Álvarez-Farizo et al., 2016)[111] or from visitors to different public areas of Doñana National and Natural Park (NPA), which is located in the southwestern Spain (Martín-López et al., 2007a,b)[112]. Overall, net family income in our sample approaches the upper bound obtained by the other studies. This may be because income in Andalusia is lower than in the rest of Spain.

Table 5.4. Socioeconomic and Attitudinal Characteristics of the contingent valuation sample

Variables	Contingent Valuation sample	
	Mean	n
Personal satisfaction when protecting the species	4.39 (0.89)	750
Personal satisfaction when contributing for a good cause	3.38 (1.37)	750
Age	38.68 (14.00)	733
Net family income (€ per month)	1,909.09 (902.58)	594
Family members	2.88 (1.35)	717
Number of members in the family that bring in income	1.73 (0.70)	713
Marital status (1=Single, 2=Married, 3=Divorced, 4=Widower)	1.67 (0.74)	749
Education (1=No schooling, 2=Primary studies, 3=Secondary school, 4=Diploma, 5=Bachelor, 6=Others)	3.43 (1.64)	749
Occupation (1=Salaried employee, 2=Entrepreneur and self-employed worker, 3=Unemployed, 4=Household tasks, 5=Student, 6=Retired, 7=Others)	2.13 (1.99)	748
Work in the forestry sector (1=Yes, 0=Otherwise)	0.05 (0.23)	750

[111] While the population sample in Castro et al. (2011) include local residents and tourist, results for socio-cultural characteristics distinguish between the two groups, so we use the first group for comparison.

[112] In the latter case, for example, each of these areas offer different results for various reasons – socio-politically and ecologically – related.

5.3.2. Payment vehicles

The model including four payment vehicles shows that they are not significant for the single-bounded and double-bounded models (Table 5.5). This means that respondents do not perceive them differently when deciding whether to pay or not. Only the variables *Intercept* and *Bid* are significant. The parameter for payment vehicle *Fund1* is negative and parameters for the rest of payment vehicles are positive, albeit close to zero. We also find that parameters are larger for the double-bounded model than for the single-bounded model, with the exception of the variable *Bid*.

Table 5.5. Single-bounded and double-bounded models using four payment vehicles

Attribute	Single-bounded	Double-bounded
	Parameter	Parameter
Intercept	2.9776*** (0.3050)	3.9990*** (0.2906)
Fund1	-0.1347 (0.2679)	-0.0795 (0.2175)
Fund2	0.1240 (0.2798)	0.1458 (0.2324)
Tax1	0.0089 (0.2856)	0.0971 (0.2294)
Bid	-0.0791*** (0.0085)	-0.1126*** (0.0634)
n	596	596
Log-likelihood	-309.73	-624.13
Adj. Mc Fadden ρ^2	0.1394	

Note: We use a dummy-coding where the reference level is Tax 2. Standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.

The WTP values reveal that differences between the four payment vehicles are not relevant. The complete combinatorial test also indicates that we cannot reject the hypothesis of statistically indistinguishable WTP in all cases (H_E, H_F, H_G, H_H, H_I and H_J) (Table 5.6).

All the same, the higher WTP is reached by the contribution to a guarantee fund, followed by increased income tax, referendum for an increased income tax and contribution to a fund, respectively.

Table 5.6. Mean willingness to pay (WTP) values from the single-bounded and double-bounded models using four payment vehicles

Payment vehicle	Single-bounded	Double-bounded
	WTP	WTP
Fund1	36.00 [31.79 – 40.65]	34.79 [32.55 – 37.09]
Fund2	39.21 [34.69 – 44.16]	36.85 [34.28 – 39.35]
Tax1	38.83 [34.32 – 43.76]	36.38 [34.01 – 38.70]
Tax2	37.56 [33.41– 42.75]	35.59 [33.05 – 37.88]
Complete combinatorial test	*p-value*	*p-value*
H_E: $WTP_{Fund1} = WTP_{Fund2}$	0.198	0.156
H_F: $WTP_{Fund1} = WTP_{Tax1}$	0.229	0.209
H_G: $WTP_{Fund1} = WTP_{Tax2}$	0.344	0.339
H_H: $WTP_{Fund2} = WTP_{Tax1}$	0.463	0.412
H_I: $WTP_{Fund2} = WTP_{Tax2}$	0.333	0.279
H_J: $WTP_{Tax1} = WTP_{Tax2}$	0.369	0.356

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets.

When payment vehicles are grouped into two categories, the variable *Fund* is not significant in any case (Table 5.7). Only the variables *Intercept* and *Bid* are significant and slightly larger than in the scenario with four payment vehicles.

Table 5.7. Single-bounded and double-bounded models using two payment vehicles

Attribute	Single-bounded	Double-bounded
	Mean parameter	Mean parameter
Intercept	3.0036***	4.0361***
	(0.2761)	(0.2708)
Fund	-0.0541	-0.0192
	(0.1979)	(0.1613)
Bid	-0.0786***	-0.1124***
	(0.0084)	(0.0063)
n	596	596
Log-likelihood	-310.22	-624.68
Adj. Mc Fadden ρ^2	0.1381	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.

The WTP values reveal that differences between the two payment vehicles have narrowed, while contribution to a fund is preferred over increased income tax. The complete combinatorial test also indicates that we cannot reject the hypothesis (H_K) that payment vehicles derive similar WTP estimates (Table 5.8). The double-bounded model offers lower WTP values compared to the single-bounded model.

Overall, our results show that the four payment vehicles do not differ from one another. Similarly, there are no differences when including only two payment vehicles. Thus, in the following analyses we work with a data-enriched model that pools the payment vehicles datasets. That is, we do not differentiate between payment vehicles, treating all of them as having the same effect on WTP.

Table 5.8. Mean willingness to pay (WTP) values from the single-bounded and double-bounded models using two payment vehicles

Payment vehicle	Single-bounded	Double-bounded
	WTP	WTP
Fund	37.68	36.21
	[34.38 – 41.53]	[32.37 – 39.83]
Tax	38.23	35.96
	[35.07 – 42.00]	[34.21 – 37.61]
Complete combinatorial test	*p-value*	*p-value*
H_K:$WTP_{Fund}=WTP_{Tax}$	0.369	0.356

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets.

Our estimates of payment vehicle effects differ from previous papers comparing payment vehicles in contingent valuation studies, which found mostly divergences. The only exception can be found in Kontoleon et al. (2005). They find that there are no differences between a product tax and a tax reallocation. Most of the papers mentioned in section 5.1.1 conclude that respondents tend to prefer taxes rather than donations or entrance fees. The only paper including referendum as payment vehicle finds that it offers the higher value (Champ et al., 2002). Still, it is difficult to make consistent comparisons between these papers and our results, as almost none of them conduct surveys in Spain, with the exception of Campos et al. (2007). However, this study focuses on forest recreation and not on biodiversity conservation. The remaining papers valuing biodiversity preservation or a particular species (Johansson, 1996; Jakobsson and Dragun, 2001; Aoun, 2015) do not value the Iberian Lynx.

5.3.3. Multiple preference orderings for the conservation of the Iberian Lynx

Models show that the parameters from the single-bounded and double-bounded models are all significant, with the exception of the variable *Homo* in the model HEP and the variable HP_{sh} in the model HSSFR (Table 5.9). The former result implies that the variable *Homo* is no longer significant when there is no strategic behavior. As expected, the variable *Bid* offers a negative sign, which means that the probability of participating in the program decreases for higher payments. When comparing the variables HP_{so}, HP_{sh} and *FR*, results indicate that, contrary to what theory predicts (Nyborg, 2000), HP_{sh} offers a higher value than HP_{so} and *FR*, respectively. The value of *Intercept* is larger in the model HSSFR than in the model HEPFR. This may be caused by the fact that it collects those individuals who behave as *HE*, while it does not take into account those individuals with a strategic behavior. The double-bounded model is better adjusted than the single-bounded model.

Table 5.9. Models HEPFR, HEP and HSSFR using the single-bounded and double-bounded models

Attribute	MODEL HEPFR		MODEL HEP		MODEL HSSFR	
	Single-bounded	Double-bounded	Single-bounded	Double-bounded	Single-bounded	Double-bounded
	Parameter	Parameter	Parameter	Parameter	Parameter	Parameter
Intercept	2.8017***	2.9964***	2.9768***	3.1297***	3.2398***	3.3898***
	(0.2701)	(0.1925)	(0.2972)	(0.2116)	(0.2932)	(0.2067)
Homo	0.3784*	0.4300**	0.2306	0.2653		
(*Homo Economicus* = 1, *Homo Politicus* = 0)	(0.2005)	(0.1526)	(0.2174)	(0.1630)		
Homo Politicus with sole responsibility (HP_{so})					-0.4379*	0.4439**
(=1 and 0 otherwise)					(0.2481)	(0.1934)
Homo Politicus with shared responsibility (HP_{sh})					0.0601	-0.0233
(=1 and 0 otherwise)					(0.2817)	(0.2075)
Homo Strategicus (FR)					-0.7839**	0.5445**
(=1 and 0 otherwise)					(0.3045)	(0.2406)
Bid	-0.0784***	-0.0951***	-0.0790***	-0.0954***	-0.0802***	0.0953***
	(0.0084)	(0.0046)	(0.0090)	(0.0050)	(0.0086)	(0.0046)
n	596	596	522	522	596	596
Log-likelihood	-308.46	-732.49	-262.84	-637.63	-305.57	-731.84
Adj. McFadden ρ^2	0.1430		0.1478		0.1510	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

As regards the WTP values, there are significant differences between types of individuals (Table 5.10). *FR* is willing to pay the lowest value and, contrary to what might be expected, HP_{so} offers the next lower value, followed by *HP*. This is true for the single-bounded and double-bounded models. *HE* and HP_{sh} provide the highest values, although the order of listing them varies depending on whether we consider the single-bounded model or the double-bounded model. In any case, differences between the two types of individuals are minimal. The double-bounded model gives lower values in the three models, while the difference for *FR* is practically non-existent.

Table 5.10. Mean willingness to pay (WTP) values from the single-bounded and double-bounded models using the models HEPFR, HEP and HSSFR

Attribute	Single-bounded	Double-bounded
	Mean WTP	Mean WTP
Model HEPFR		
Homo Economicus (HE)	40.77 [37.04 – 45.24]	36.02 [34.08 – 38.04]
Homo Politicus (HP)	35.74 [32.87 – 39.05]	31.57 [29.66 – 33.43]
Model HEP		
Homo Economicus (HE)	40.79 [36.83 – 45.56]	35.54 [33.53 – 37.70]
Homo Politicus (HP)	37.67 [34.54 – 41.38]	32.84 [30.79 – 34.80]
Model HSSFR		
Homo Politicus with sole responsibility (HP_{so})	34.99 [30.92 – 39.28]	30.90 [28.26 – 33.65]
Homo Politicus with shared responsibility (HP_{sh})	41.15 [36.23– 46.67]	35.38 [32.33 – 38.40]
Homo Strategicus (FR)	30.64 [25.65 – 35.84]	29.88 [26.34 – 33.29]

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

What is relevant in this context, as compared with Nyborg (2000), is the existence of free-riding. This leads us to ask about the role that the strategic behavior adopts when comparing the dichotomy HE/HP between the model HEPFR and the model HEP. When the free-riding effect is eliminated, WTP for *HP* is larger than before, while the effect over WTP for *HE* is less limited or even nil. Therefore, the WTP value for these two types of individuals move significantly closer to each other. This is because *HP* is more prone to strategic behavior than *HE*.

The complete combinatorial test indicates that we cannot reject the hypothesis of statistically indistinguishable WTP between *HE* and *HP* in the model HEPFR in the single-bounded model (H_M) and between HP_{so} and HP_{FR} in the model HSSFR single-bounded and double-bounded models (H_P). However, this is rejected for the rest of hypothesis (see Table 5.11 and Table 5.12).

Table 5.11. Complete combinatorial test result from the single-bounded and double-bounded models using models HEPFR, HEP and HSSFR

	Single-bounded *(p value)*	Double-bounded *(p value)*
Model HEPFR		
H_L: $WTP_{HE} = WTP_{HP}$	0.049**	0.003***
Model HEP		
H_M: $WTP_{HE} = WTP_{HP}$	0.173	0.056*
Model HSSFR		
H_N: $WTP_{HPsh} = WTP_{FR}$	0.008***	0.023**
H_P: $WTP_{HPso} = WTP_{FR}$	0.136	0.349
H_Q: $WTP_{HPso} = WTP_{HPsh}$	0.057*	0.032**

Note: standard errors are shown in brackets; N: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

In short, our results reveal that respondents who behave strategically are willing to pay less than the rest of individuals, which is a logical result. By contrast, individuals identified as *HE* and HP_{sh} provide the highest WTP values, and, contrary to expectations, respondents defined as HP_{sh} offers a larger WTP than HP_{so}. We also find that ‘citizen’ behavior is more likely to have a strategic behavior than the ‘consumer’ behavior.

As there is no data available that offer information on the mean WTP values for the conservation of the Iberian Lynx, we may compare the WTP values from Table 5.10 with the values estimated in recent studies valuing biodiversity conservation or a set of species in Spain and in other countries. Martín-López et al. (2007a,b) offer a mean WTP of €28.03

and €29.70 (updated to 2008 Euros), respectively, for biodiversity conservation services in the Doñana NPA, so our estimations are in the upper bound of this range. Martín-López et al. (2007b) take into account the financial contribution that different users would be willing to pay for the conservation of 15 species, including the Iberian Lynx. They also analyze the attitude of respondents towards these species and the economic value placed on each species. Results show that the highest value is attached to the Iberian Lynx.

Other studies have been conducted in different countries in order to estimate the WTP for specific species, such as the Eurasian otter (*Lutra lutra*) in Britain, Leadbeater's possum (*Gymnobelideus leadbeateri*) in Australia and Gray wolf (*Canis lupus*) in the United States (White et al., 1997; Jakobsson and Dragun, 2001; Chambers and Whitehead, 2003). They offer a mean WTP between €18.23 and €52.92 (updated to 2008 Euros), so our estimations are within the range of values offered by these studies. In particular, Jakobsson and Dragun (2001) estimate WTP values for the Leadbeater's possum in Australia, differentiating between two payment vehicles: tax and donation. They offer a WTP mean of €18.23 and €52.92, respectively (updated to 2008 Euros). White et al. (1997) provide a WTP value of 20.77 for the conservation of the Eurasian otter in Britain (updated to 2008 Euros). Chambers and Whitehead (2003) evaluate a plan for protecting the Gray wolf in Minnesota, conducting surveys in two locations: one is located on the heart of the wolf habitat, while the other is outside of this area. They find that 'locals' are willing to pay a lower amount (€6.23) in comparison with residents living far away from the area of habitat for the gray wolf (€28.97) (updated to 2008 Euros). An interesting study for a protection program of the Alaskan Steller sea lion (*Eumetopias jubatus*) can be found in Giraud et al. (2004). However, we have not included it in our comparison, as it offers a very wide range, depending on the sample chosen (from €26.82 to €105.56 updated to 2008 Euros).

5.3.4. Results about the clarity, credibility and certainty of the scenario

Before presenting the results for the models HEPFR, HEP and HSSFR, including three new explanatory variables (*Clarity*, *Credibility* and *Certainty*), let us look at the number of respondents answering to each of these three questions (Table 5.12).

Table 5.12. Number of respondents answering questions related to the clarity, credibility and certainty of the contingent valuation scenario

	1	2	3	4	5	n	Mean
Clarity question							
Homo Economicus (HE) including FR	4	9	32	88	146	279	4.30
Homo Politicus (HP) including FR	6	14	59	134	104	317	3.99
Homo Economicus (HE) excluding FR	4	8	29	77	129	247	4.29
Homo Politicus (HP) excluding FR	3	13	54	110	95	275	4.02
Homo Politicus with sole responsibility (HP_{so})	2	4	32	65	52	155	4.04
Homo Politicus with shared responsibility (HP_{sh})	1	9	22	45	43	120	4.00
Homo Strategicus (FR)	3	2	8	35	26	74	4.07
Credibility question							
Homo Economicus (HE) including FR	44	78	100	37	20	279	2.68
Homo Politicus (HP) including FR	51	105	100	39	22	317	2.61
Homo Economicus (HE) excluding FR	41	71	92	26	17	247	2.62
Homo Politicus (HP) excluding FR	43	87	88	35	22	275	2.66
Homo Politicus with sole responsibility (HP_{so})	32	46	52	16	9	155	2.51
Homo Politicus with shared responsibility (HP_{sh})	11	41	36	19	13	120	2.85
Homo Strategicus (FR)	11	25	20	15	3	74	2.65
Certainty question							
Homo Economicus (HE) including FR	3	3	27	66	180	279	4.49
Homo Politicus (HP) including FR	5	8	38	137	129	317	4.19
Homo Economicus (HE) excluding FR	3	3	26	58	157	247	4.47
Homo Politicus (HP) excluding FR	5	6	33	125	106	275	4.17
Homo Politicus with sole responsibility (HP_{so})	5	3	16	74	57	155	4.13
Homo Politicus with shared responsibility (HP_{sh})	0	3	17	51	49	120	4.22
Homo Strategicus (FR)	0	2	6	20	46	74	4.49

Note: n indicates the number of observations

As shown, the majority of respondents found the contingent valuation question very clear, did not doubt very much when answering it and believed that it was not plausible that the program could be implemented using these payment vehicles. When differentiating between types of individuals, *HE* and *FR* were the most confident and found the contingent valuation question clearer. *HE* is also more convinced than *HP*. The latter was confident, albeit less than the rest of individuals and found the question a little bit complicated. In the case of HP_{so} and HP_{sh}, there are hardly any differences between them and with *HP* with respect to the credibility question, while HP_{sh} is more convinced than HP_{so}.

Concerning the models HEPFR, HEP and HSSFR, including the *Clarity*, *Credibility* and *Certainty* questions, the parameters from the single-bounded and double-bounded models are all significant, except for the variable *Homo* in the model HEP in the single-bounded and double-bounded models, the variable HP_{sh} in the model HSSFR in the single-bounded and double-bounded models, the variable HP_{so} in the model HSSFR for the single-bounded model, and the *Clarity* and *Certainty* questions in all cases (Table 5.13). The variable *Bid* offers a negative sign, and parameters for the *Credibility* question are higher in comparison to the ones for the other two questions. The parameters of the double-bounded model are larger than the parameters of the single-bounded model.

Table 5.13. Models HEPFR, HEP and HSSFR using the single-bounded and double-bounded models and questions related to the clarity, credibility and certainty of the contingent valuation scenario

	Model HEPFR		Model HEP		Model HSSFR	
Attribute	Single-bounded	Double-bounded	Single-bounded	Double-bounded	Single-bounded	Double-bounded
	Parameter	Parameter	Parameter	Parameter	Parameter	Parameter
Intercept	2.2651***	2.4109***	2.3695***	2.6506***	2.6370***	2.9171***
	(0.6496)	(0.4373)	(0.6927)	(0.4599)	(0.6615)	(0.4627)
Homo	0.3497*	0.4107**	0.2250	0.2663		
(*Homo Economicus* = 1, *Homo Politicus* = 0)	(0.2027)	(0.1544)	(0.2198)	(0.1645)		
Homo Politicus with sole responsibility (HP_{so})					-0.4120	-0.4484**
(=1 and 0 otherwise)					(0.2509)	(0.1955)
Homo Politicus with shared responsibility (HP_{sh})					0.0405	-0.0193
(=1 and 0 otherwise)					(0.2850)	(0.2089)
Homo Strategicus (FR)					-0.7799*	-0.5431**
(=1 and 0 otherwise)					(0.3073)	(0.2410)
Clarity	0.0988	0.0654	0.2343	0.2982	0.1578	0.2187
(=1 if answer was 3,4 or 5 and 0 otherwise)	(0.4372)	(0.3334)	(0.4774)	(0.3643)	(0.4354)	(0.3226)
Credibility	0.6196**	0.4144**	0.6518**	0.4130**	0.6124**	0.4154**
(=1 if answer was 3,4 or 5 and 0 otherwise)	(0.2019)	(0.1560)	(0.2120)	(0.1678)	(0.2031)	(0.1549)
Certainty	0.1516	0.3444	0.0616	0.0550	0.1436	0.2751
(=1 if answer was 3,4 or 5 and 0 otherwise)	(0.5643)	(0.3819)	(0.6126)	(0.4071)	(0.5657)	(0.3979)
Bid	-0.0784***	-0.0957***	0.0789***	-0.0962***	-0.0800***	-0.0953***
	(0.0086)	(0.0047)	(0.0092)	(0.0051)	(0.0087)	(0.0047)
n	596	596	522	522	596	596
Log-likelihood	-303.49	-728.51	-258.02	-633.69	-300.67	-731.35
Adj. McFadden ρ^2	0.1568		0.1634		0.1646	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g., ***, **, *) denote significance at the 1%, 5% and 10% level, respectively. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

Previously to introduce the results for the WTP values, let us explain how these values have been estimated for each type of individuals and each of the three questions analyzed in this section. If the clarity question took value 1 and the credibility and certainty questions took value 0, we would be assuming that only respondents who considered the contingent valuation question easy, answered it with some difficulties and though that there was a low possibility of implementing the program were included[113]. As it would not make much sense to estimate WTP in this way, we define WTP as follows:

$$WTP_{HE(clarity)} = \frac{\beta_{INntercept}}{\beta_{Bid}} + \frac{\beta_{Homo}}{\beta_{Bid}} X_{Homo} + \frac{\beta_{clarity}}{\beta_{Bid}} X_{clarity} + \frac{\beta_{credib}}{\beta_{Bid}} \overline{X_{credib}} + \frac{\beta_{certain}}{\beta_{Bid}} \overline{X_{certain}}$$

where the clarity question takes value 1, and the credibility and certainty questions take their mean value from the sample. The same procedure has been followed for the remaining questions.

The WTP values, in general terms, are slightly higher than those estimated in Table 5.10, especially for the *Credibility* question (Tables 5.14 and 5.15). The latter result may be explained by the fact that the few respondents who found the possibility of implementing the conservation program more feasible would pay a very high amount of money. However, we may also be dealing with a hypothetical bias (Cumming et al., 1995, 1997, 1998; Bjornstad et al., 1997), as those who are not convinced that the conservation program can be put into practice may have tended to state that they are willing to pay a higher amount of money than that they would actually pay if the program were implemented in the real life. Additionally, WTP values obtained for the *Clarity* and *Certainty* questions do not differ greatly from the results of Table 5.10, above all when all individuals are included. When the free-rider is excluded, differences are a little bit higher. The single-bounded model provides larger values than the double-bounded model in the three models. These differences are larger for the model HEP and lower for the model HSSFR. Still, the difference between the two models is smaller than that in Table 5.10. Comparing between types of individuals, *HE* is willing to pay more than *HP* in all models, and the same is true for HP_{sh} in comparison with HP_{so} and *FR*. The difference between them is reduced in the double-bounded model.

[113] If the three questions took value 1, only respondents who considered the contingent valuation question easy, answered it without difficulties and though that it was possible to the program were included.

Table 5.14 Mean willingness to pay (WTP) values from the single-bounded and double-bounded models using models HEPFR and HEP and questions about the clarity, credibility and certainty of the contingent valuation scenario

WTP	Model HEPFR		Model HEP	
	Single-bounded	Double-bounded	Single-bounded	Double-bounded
Homo Economicus (HE)				
Clarity	40.78 [36.92 – 45.26]	35.98 [34.10 – 37.86]	41.13 [36.97 – 46.06]	35.77 [33.76 – 37.78]
Credibility	44.44 [39.81 – 49.63]	37.96 [35.78 – 40.19]	44.86 [41.01 – 50.60]	37.58 [35.27 – 39.98]
Certainty	40.77 [36.90 – 45.31]	36.06 [34.15 – 37.99]	41.01 [36.80 – 45.90]	35.60 [33.60 – 37.63]
Homo Politicus (HP)				
Clarity	36.33 [33.26 – 39.77]	31.63 [29.60 – 33.61]	38.29 [34.93 – 42.12]	32.93 [30.78 – 35.06]
Credibility	39.99 [36.13 – 44.38]	33.60 [31.25 – 35.92]	42.02 [37.83 – 46.83]	34.74 [32.26 – 37.19]
Certainty	36.32 [33.19 – 39.69]	31.70 [29.67 – 33.66]	38.16 [34.73 – 41.85]	32.76 [30.59 – 34.87]

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect.

Table 5.15. Mean willingness to pay (WTP) values from the single-bounded and double-bounded models using the model HSSFR and questions about the clarity, credibility and certainty of the contingent valuation scenario

WTP	Model HSSFR	
	Single-bounded	Double-bounded
Homo Politicus with sole responsibility (HP_{so}*)*		
Clarity	36.99 [32.58 – 41.29]	34.22 [30.82 – 37.71]
Credibility	39.03 [34.03 – 43.96]	35.35 [31.48 – 39.26]
Certainty	36.96 [32.67 – 41.35]	34.21 [30.78 – 37.70]
Homo Politicus with shared responsibility (HP_{sh})		
Clarity	42.73 [37.55 – 48.38]	38.73 [35.16 – 42.44]
Credibility	44.77 [39.40 – 50.72]	39.86 [35.90 – 44.07]
Certainty	42.07 [37.67 – 48.43]	38.72 [35.19 – 42.34]
Homo Strategicus (FR)		
Clarity	32.39 [26.85 – 38.05]	33.18 [29.15 – 37.15]
Credibility	34.43 [28.63 – 40.24]	34.32 [29.92 – 38.66]
Certainty	32.36 [26.87 – 38.00]	33.17 [29.19 – 37.14]

Note: lower and upper bounds of the confidence interval (95%) are shown in brackets. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

The complete combinatorial test indicates that we cannot reject the hypothesis of statistically indistinguishable WTP: (i) between *HE* and *HP* for the *Credibility* question in the model HEPFR and for the three questions in the model HEP in the single-bounded model (H_R), (ii) between HP_{so} and HP_{FR} for the three questions in the single-bounded and double-bounded models (H_T), and (iii) between HP_{so} and HP_{sh} for the *Credibility* question in the single-bounded model (H_U). However, this is rejected for the rest of hypothesis (see Table 5.16 and Table 5.17).

Table 5.16. Complete combinatorial test result from the single-bounded and double-bounded models using models HEPFR and HEP and questions about the clarity, credibility and certainty of the contingent valuation scenario

WTP	Model HEPFR		Model HEP	
	Single-bounded	Double-bounded	Single-bounded	Double-bounded
H_R:$WTP_{HE} = WTP_{HP}$				
Clarity	0.079*	0.004***	0.207	0.056*
Credibility	0.127	0.011**	0.253	0.082*
Certainty	0.079*	0.005***	0.206	0.056*

Note: standard errors are shown in brackets; N: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively. HEPFR refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*. HEP refers to the model that differentiates between *Homo Economicus* and *Homo Politicus*, while excluding the free-riding effect.

Table 5.17. Complete combinatorial test result from the single-bounded and double-bounded models using the model HSSFR and questions about the clarity, credibility and certainty of the contingent valuation scenario

WTP	Model HSSFR	
	Single-bounded	Double-bounded
H_S:$WTP_{HPsh} = WTP_{FR}$		
Clarity	0.012**	0.045**
Credibility	0.017**	0.062*
Certainty	0.011**	0.044**
H_T: $WTP_{HPso} = WTP_{FR}$		
Clarity	0.143	0.380
Credibility	0.163	0.393
Certainty	0.144	0.381
H_U: $WTP_{HPso} = WTP_{HPsh}$		
Clarity	0.084*	0.063*
Credibility	0.104	0.088*
Certainty	0.084*	0.063*

Note: standard errors are shown in brackets; N: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively. HSSFR refers to the model that differentiates between *Homo Economicus*, *Homo Politicus with shared responsibility*, *Homo Politicus with sole responsibility* and *Homo Strategicus*.

Overall, our results show that there are differences between *HE* and *HP* when all respondents – including *FR* – are considered. However, there is less divergence between these two types of individuals when taking out the strategic behavior. When it comes to the WTP values, the highest value is obtained for the *Credibility* question in regard to *Clarity* and *Certainty* questions.

5.3.5. Aggregated values

For aggregation purposes, we estimate the number of individuals involved in the program for the conservation of the Iberian Lynx by considering the population from each of the provinces in our stratified sample[114]. The Spanish Statistical Office estimates 15,420,870 inhabitants in 2008 for these provinces.

To obtain aggregated values (in 2008 Euros), we work with models HEPFR and HSSFR, as the variable *Homo* is only significant in these two models. Results reveal that the single-bounded model offers higher mean WTP and aggregated values for the *CV* calculation than the double-bounded model in all cases (Table 5.18). The resulting price for the *SEV* scenario, p^*, and aggregated values are lower than mean WTP and aggregated values for the *CV* calculation. As shown, the percentage of individuals who are willing to take part in the program is clearly lower for the *SEV* calculation. Specifically, if all respondents were considered as *HE*, only 62.60% of them would be willing to participate and therefore to pay p^*. In the case of *HP*, this percentage dropped to 59.20%. For HP_{so}, HP_{sh} and *FR*, percentages are 58.80%, 63.95% and 55.24%, respectively (we shall return to this issue in due course).

If we compare the five types of individuals, HP_{sh} offers the higher aggregated value for the *CV* and *SEV* calculations in the single-bounded model. This is not the case in the double-bounded model, as *HE* offers the highest aggregated value for the two measures. Either way, values for these two types of respondents are very similar. As expected, *FR* provides the lowest values in all cases.

[114] We realize that payment vehicles have been described in a hypothesis of fund raising so that it would be possible to collect the entire consumer surplus through the establishment of two payment vehicles: increased income tax and contribution to a fund. However, for simplicity and because it is clearly a more realistic scenario, in this case we do not differentiate between these payment vehicles and assume that a ‘price’ to be compulsory paid to conserve the Iberian Lynx is set.

Table 5.18. Aggregated values of compensating variation (*CV*) and simulated exchange value (*SEV*) of a conservation program for the Iberian Lynx in Spain (year 2008)

Measure	Single-bounded			Double-bounded		
	€ per person	Population	Aggregated €	€ per person	Population	Aggregated €
Compensating Variation						
Homo Economicus	40.57	15,420,870	625,624,696	36.03	15,420,870	555,613,946
Homo Politicus	35.75	15,420,870	551,296,102	31.51	15,420,870	485,911,614
With sole responsibility	34.94	15,420,870	538,805,198	30.91	15,420,870	476,659,092
With shared responsibility	41.15	15,420,870	634,568,800	35.33	15,420,870	544,819,337
Strategicus	30,62	15,420,870	472,187,039	29.86	15,420,870	460,467,178
Simulated Exchange Value						
Homo Economicus	34.00	9,653,464	328,217,776	29.50	10,029,734	295,877,153
Homo Politicus	31.00	9,129,155	283,003,805	27.00	9,338,879	252,149,733
With sole responsibility	30.50	9,067,471	276,557,865	26.50	9,308,037	246,662,980
With shared responsibility	34.00	9,861,646	335,295,964	29.00	9,966,508	289,028,732
Strategicus	28.00	8,518,488	238,517,664	26.00	9,112,192	236,916,992

Coming back to Table 5.13, results indicate that only the variable *Credibility* is significant. As the potential population taking part in our valuation scenario is better represented by those individuals who really think that it is possible to carry out this program, we use this question in order to know whether results change with respect to the previous scenario.

Results shown that mean WTP and p^* are higher than average values in Table 5.18 in all cases (Table 5.19). Still, differences are slightly higher in the single-bounded model than in the double-bounded model and for the *CV* measure. The percentage of individuals participating in the program under the *SEV* measure is also larger than in the previous scenario for the single-bounded and double-bounded models. However, if each type of respondent represented the whole population, the percentage of *HE* who would be willing to participate would rise to 65.67%. For *HP*, HP_{so}, HP_{sh} and *FR*, percentages are 60.39%, 61.94%, 65.75% and 58.88%, respectively. These percentages are higher than in the previous case. Thus, aggregated values are also higher than before.

Table 5.19. Aggregated values of compensating variation (*CV*) and simulated exchange value (*SEV*) of a conservation program for the Iberian Lynx in Spain (year 2008) using the question related to the credibility of the contingent valuation question

Measure	Single-bounded			Double-bounded		
	€ per person	Population	Aggregated €	€ per person	Population	Aggregated €
Compensating Variation						
Homo Economicus	44.31	15,420,870	683,298,750	37.97	15,420,870	585,530,434
Homo Politicus	39.82	15,420,870	614,059,043	33.65	15,420,870	518,912,275
With sole responsibility	39.06	15,420,870	602,339,182	33.11	15,420,870	510,585,006
With shared responsibility	44.62	15,420,870	688,079,219	37.13	15,420,870	572,576,903
Strategicus	34.47	15,420,870	531,557,389	31.95	15,420,870	492,696,796
Simulated Exchange Value						
Homo Economicus	36.50	10,126,885	369,631,302	30.50	10,307,309	319,526,579
Homo Politicus	36.50	9,312,663	339,912,199	30.50	9,360,468	290,174,508
With sole responsibility	33.00	9,551,687	315,205,671	28.00	9,838,515	290,236,192
With shared responsibility	36.50	10,139,222	370,081,603	30.50	10, 336,609	335,939,792
Strategicus	30.00	9,079,808	272,394,240	27.00	9,840,057	280,441,624

In a nutshell, market simulation implies lower aggregated values for implementing the conservation program for the Iberian Lynx for the single-bounded and double-bounded models and for different types of individuals.

As explained in the fourth chapter, our estimations must be taken carefully, as the survey was conducted in 2008, so the estimated economic values may have varied.

5.4. Intermediate summary

In this chapter, we have presented a contingent valuation study for valuing multiple preference orderings in a biodiversity conservation context. We have gone beyond the model set out by Nyborg (2000) due to the inclusion of the free-rider problem.

Based on this survey, and considering that payment vehicles are not significant, we have first built a pooled model to differentiate between two types of individuals: *Homo Economicus* and *Homo Politicus*. According to this model, and, as expected, the former is willing to contribute with a larger amount of money than the latter when it comes to implementing a program to conserve the Iberian Lynx. *Homo Economicus* is more committed to supporting it and it is not concerned with the attitude of the rest of individuals. By contrast, the behavior of *Homo Politicus* considers the program necessary but at the same time it cares about what other respondent do.

We also identify individuals who behave strategically within these two types of respondents. Thus, if we eliminate this effect from the previous dichotomy *Homo Economicus*/*Homo Politicus*, results show that *Homo Politicus* tends more towards a strategic behavior than *Homo Economicus*. While the WTP value for *Homo Economicus* changes almost nothing when excluding the free-rider problem, *Homo Politicus* offers a clearly higher WTP, so WTP values for both types of individuals converge more.

In the third model presented in this chapter we have divided Homo *Politicus* into *Homo Politicus with sole responsibility*, *Homo Politicus with shared responsibility* and *Homo Strategicus*. Contrary to the theoretical result pointed out by Nyborg (2000), our model indicates that respondents considered as *Homo Politicus with shared responsibility* are willing to pay a higher amount of money to conserve the Iberian Lynx. The WTP value for the latter individual is, indeed, similar to that obtained for *Homo Economicus*, especially when free-riding is eliminated.

We also analyze whether respondents found the contingent valuation easy, the certainty with which they answered it and the credibility of the conservation program. There are an important number of respondents who think that it is unlikely to launch the above program, but instead results have indicated that WTP values for the credibility question are higher than those for questions inherent to the conduct of a survey, such as the clarity and certainty questions. This result may be tackled in a two-fold manner: on the one hand, the WTP of those respondents who are really convinced is very high. On the other hand, those who think that the implementation of the conservation program is unrealistic may have offered a higher hypothetical WTP than what they would actually be willing to pay. The fact that they consider the possibility of conducting the program as being somewhat remote may lead them to think that it is a good idea to protect the

Iberian Lynx, but that they will not have to pay for this. In any case, both options are not contradictory, as the two attitudes point in the same direction: most of respondents – both those who consider possible to carry out the program as well as those who do not – find the program to conserve the Iberian Lynx in Spain an interesting proposal.

Concerning the aggregated values, our results confirm that *CV* estimates offers higher values than the *SEV* method. Furthermore, under a scenario of multiple preferences ordering in an environmental valuation exercise, differences between types of individuals are larger with the *CV* estimate than with the *SEV* method. In other words, there is closer convergence between types of individuals when using the simulated scenario, so the *SEV* measure minimizes the bias of distinct preference. By contrast, the different roles adopted by individuals affect welfare measures most. Moreover, the fact of not integrating the free-rider problem, as other papers do, may make aggregated values be overestimated or underestimated, so this effect is relevant and should be borne in mind, especially in the valuation of public goods such as the one analyzed in this chapter.

CHAPTER 6. CONCLUSIONS/CONCLUSIONES

6. CONCLUSIONS

The research work developed in this dissertation has mainly involved analyzing how non-market ecosystem services can be incorporated into the national accounts in a consistent way with market goods and services.

As explained throughout the dissertation, there are different perspectives from which this issue has been pursued: accounting, theoretical and valuation. However, we have shown that there is a gap between them when it comes to valuing ecosystem goods and services without observable market prices. Hence, one primary goal of the present reseach has been to bring these differing viewpoints together under a common methodology: the Simulated Exchange Value. We have combined a theoretical optimal control model with two applied studies focused on the valuation of the recreational use in two types of Spanish forests, on the one hand, and on the conservation of the Iberian Lynx in Spain, on the other hand. This combination has proven to be successful for including non-market ecosystem services in the national accounts consistently with the rest of goods and services, regardless of the technique applied.

From a theoretical point of view, we have set out the main findings for the forestry sector obtained by the theoretical literature on green national accounting. In this regard, we have shown that non-market ecosystem services can be integrated into the Net National Product calculation at their (*simulated*) exchange values. This result is perfectly consistent with the proposals of the national accounting systems, as the latter indicate that valuation should be based on exchange values whenever possible.

Specifically, we have developed an optimal control model in which the utility function to be maximized does not depend entirely on consumption of market goods and services, but also on services provided by forests because they contribute to current and future well-being of individuals. In order to be able to estimate the value that these services would have if they were incorporated into national accounting, we have employed the Simulated Exchange Value method. We have focused on forest recreational use and assumed that the payment of an entrance fee to gain access to the forest would be set. In this way, this payment would be equivalent to the price that would be paid for consuming a market good or service. The demand of the aforementioned service would therefore depend on the price of such entrance fee and the offer would be determined by the number of visitors to whom the forest owner permits accessing. In any case, this method could be applied to any other services by following the same procedure, which represents an advantage over other methodologies.

From an empirical point of view, we have demonstrated that the Simulated Exchange Value method can be equally applied in order to calculate monetary values of public recreation in Spanish stone pine and cork oak forests, on the one hand, and of the conservation of the Iberian Lynx in Spain, on the other. As this method requires the estimation of the demand function, we have employed a choice experiment and a contingent valuation survey, respectively, to do so. In spite of the fact that we have used different tools from those of the theoretical approach, the application of the Simulated Exchange Value method remains valid and affords a series of advantages over the welfare measures commonly used by most of studies within the cost-benefit analysis.

In particular, we have compared compensating variation (a type of Hicksian variation) with the results obtained when applying the SEV method in order to obtain the aggregated economic values for both services. The two measures rely on the assumptions about the demand curve (WTP distribution) implicit in the discrete choice model used. However, while the simulated exchange value uses only a specific area under the demand function, the compensating variation covers the whole area under this function and is therefore more sensitive to changes to the assumptions about its shape. Meanwhile, the simulated exchange value approach is sensitive to the inclusion of substitute alternatives in the simulated market as this conditions the probability function. This is not the case for the compensating variation estimate, which directly uses the estimated parameters of the utility function.

Another relevant implication of the Simulated Exchange Value method, especially in the case of forest recreational use, is that the simulated market would have distributional consequences on current visitors. The relative effort (in monetary terms) needed to keep accessing the forest will be different depending on the characteristics of the visitors and their families. In fact, a proportion of current visitors will be left out of the forest as they would not be willing to pay the price set. This type of analysis would require additional models with explanatory variables that allow us to identify the characteristics of those respondents that are more affected by the simulated market. However, this analysis goes beyond the scope of our dissertation, although we acknowledge its interest for future research.

In addition to the above-mentioned, the study on forest recreational use has addressed the issue concering the validity of choice experiments. We have conducted two convergent validity tests. On the one hand, we have obtained similar results from a choice and from a ranking recoded as a choice in an exercise involving three alternatives plus the status quo. This reinforces the idea that people make consistent choices in these experiments and it allows the practitioner to use ranking formats, with the knowledge that the first rank can be analyzed as a standard choice and that additional information from subsequent ranks can be used. On the other hand, we have ascertained that respondents are willing to pay €2.7 more as increased trip expenditures for each additional euro paid as an entrance fee. While these additional payments would have the same effect on the disposable income of forest recreationists, they do not seem to have the same effect on their utility. We believe that these differences may be explained by the perception of some forest recreationists that they have the right to freely access the forest and by the strategic behavior of some respondents who try to avoid the establishment of fees for accessing current free access areas.

As regards the study on the conservation of the Iberian Lynx in Spain, we have analyzed the role adopted by different respondents when facing a payment scenario to allow the conservation program to be implemented. Given that we are dealing with a public good, not all individuals act as consumers, but can judge this matter from society’s point of view. In our study, we extend this dichotomy in order to take into account a new type of individual who stands out for his/her strategic behavior. In fact, this behavior is key to better understanding the preferences adopted. As expected, they are the ones who are willing to pay the lower amount to preserve the Iberian Lynx. Likewise, results show that most of respondents though that it was not plausible that the program could be implemented. This may lead us to conclude that it is relevant to make efforts to provide scenarios that may be understood as credible. Otherwise, we might face hypothetical bias.

Hence, this dissertation shows that the challenge faced by non-market valuation techniques for valuing non-market ecosystem services goes beyond the application of the method itself – it also includes how to integrate willingness to pay values in an extended analysis that intends to incorporate the economic value of these services. Further research should explore additional assumptions about this integration as well as alternative scenarios that might more closely resemble how the potential market for these services would work.

Apart from the conclusions drawn so far, another important contribution of this dissertation is the treatment of net accumulation of forest capital. In accordance with previous literature, the theoretical model explained above has shown that the current-value Hamiltonian and Green Net National Product can be valued by using only values of the current year, which clearly facilitates the analysis. However, this assumption is far-fetched, as it assumes that there is a single species, among other assumptions. We have extended this model to allow different species growing differently to exist, thus showing that it is no longer possible to use only values of the current year, as Green Net National Product includes future values. To that end, we have utilized a Volterra integral equation, which had not been applied within a green national accounting framework until now. In this way we show that it is also possible to use this type of tools within this approach.

CONCLUSIONES

El trabajo de investigación desarrollado en esta tesis doctoral ha consistido fundamentalmente en el análisis de cómo los servicios de los ecosistemas de no mercado pueden ser incorporados a la contabilidad nacional de forma consistente con los bienes y servicios de mercado.

Como se ha explicado a lo largo de la tesis, existen diversas perspectivas desde las que se ha venido abordado esta cuestión: contable, teórica y de valoración. No obstante, hemos mostrado que existe una brecha entre ellas en lo que se refiere a la valoración de los bienes y servicios proporcionados por los bosques para los que no existen precios de mercado. Por lo tanto, un objetivo primordial de la presente investigación ha sido aunar estas perspectivas en torno a una metodología común: el Valor de Cambio Simulado. Para ello, hemos combinado un modelo teórico de control óptimo con dos estudios aplicados orientados a la valoración del uso recreativo en dos tipos de bosques españoles, por un lado, y de la conservación del lince Ibérico, por otro. Esta combinación ha demostrado que, independientemente de las técnicas que apliquemos, los servicios de los ecosistemas de no mercado pueden ser incluídos en la contabilidad nacional consistemententemente al resto de bienes y servicios.

Desde un punto de vista teórico, hemos presentado las principales conclusiones obtenidas para el sector forestal por la literatura teórica en torno a la contabilidad nacional verde. En este sentido, hemos mostrado que los bienes y servicios de los ecosistemas de no mercado pueden ser integrados en el cálculo del Producto Nacional Neto estimando sus valores de cambio (simulados). Este resultado es perfectamente compatible con la propuesta realizada por los sistemas de cuentas, ya que éstos indican que la valoración debe hacerse utilizando valores de cambio siempre que sea posible.

En concreto, hemos desarrollado un modelo de control óptimo en el que la función de utilidad a maximizar no depende únicamente del consumo de bienes y servicios 'de mercado' sino también de los servicios ofrecidos por los bosques dado que éstos contribuyen al bienestar presente o futuro de los individuos. Para poder estimar el valor que estos servicios tendrían si se incorposen a la contabilidad nacional, hemos empleado el Valor de Cambio Simulado. Nos hemos centrado en el uso recreativo de los bosques, asumiendo que se establecería el pago de una entrada para poder acceder al bosque. De este modo, este pago sería el equivalente al precio que se pagaría por consumir un bien o servicio comercial. Así, la demanda del mencionado servicio dependerá del precio de dicha entrada y la oferta vendrá determinada por el número de visitantes a los que el propietario del bosque permita el acceso. En cualquier caso, este método podría aplicarse a cualquier otro servicio siguiendo el mismo procedimiento, lo cual representa una ventaja respecto a otras metodologías.

Desde una perspectiva aplicada, hemos demostrado que el método del Valor de Cambio Simulado puede ser igualmente utilizado para calcular los valores monetarios del uso recreativo público en bosques de pino piñonero y alcornoque en España, por un lado, y la conservación del lince Ibérico en España, por otro. Dado que este método requiere estimar la función de demanda, hemos empleado para ello un experimento de elección y una encuesta de valoración contingente, respectivamente. Pese a que hemos utilizado otras herramientas diferentes a las del modelo teórico, se puede observar que la aplicación del método del Valor de Cambio Simulado sigue siendo válida y reporta una serie de ventajas frente a las medidas de bienestar que comúnmente utilizan la mayoría de los estudios dentro del análisis coste-beneficio.

En particular, hemos comparado la variación compensatoria (un tipo de variación Hicksiana) con los resultados obtenidos de aplicar el método de Valor de Cambio Simulado para calcular los valores agregados de ambos servicios. En cualquier caso, las dos medidas se basan en los supuestos implícitos sobre la curva de demanda (la distribución de la disposición a pagar) en los modelos de elección discreta utilizados. Sin embargo, mientras que el método de Valor de Cambio Simulado usa únicamente un área específica debajo de la función de demanda, la variación compensatoria cubre el área completa debajo de esta función y, por lo tanto, es más sensible ante cambios en los supuestos sobre su forma. Mientras tanto, el enfoque de simular valores de cambio es más sensible a la inclusión de alternativas que puedan ser sustitutivas en el mercado simulado, ya que esto condiciona la función de probabilidad. Esto no es el caso de la variación compensatoria, la cual utiliza directamente los parámetros estimados de la función de la utilidad.

Otra importante implicación del método del Valor de Cambio Simulado, especialmente para el caso del uso recreativo, es que el mercado simulado tendría consecuencias distributivas sobre los visitantes actuales. El esfuerzo relativo en términos monetarios que se necesitaría para seguir pudiendo acceder al bosque variará dependiendo de las características de los visitantes y sus familias. De hecho, una proporción de los visitantes actuales no podría disfrutar de este servicio ya que no estaría dispuesta a pagar el precio establecido. Este tipo de análisis requeriría desarrollar modelos adicionales con variables explicativas que nos permitiesen identificar las características de aquellos encuestados que se ven más afectados por el mercado simulado. No obstante, este análisis va más allá del objetivo de la tesis, si bien consideramos que sería un tema interesante a tratar en el futuro.

Además de lo ya mencionado, en el estudio aplicado al uso recreativo hemos abordado la cuestión relativa a la validez de los experimentos de elección. Para ello, hemos llevando a cabo dos análisis de validez convergente. Los resultados muestran convergencia entre un formato de elección y otro de ordenación recodificado como una elección en un experimento con tres alternativas más el status quo. La principal implicación de este resultado es que los encuestados realizan elecciones consistentes en estos experimentos y los evaluadores obtienen informacion adicional de usar un formato de ordenación sin perder los resultados de un ejercicio de elección equivalente. Por otro lado, hemos constatado que los encuestados están dispuestos a pagar 2,7 euros más como incremento en los gastos de viaje por cada euro adicional pagado como entrada. Aunque estos pagos adicionales tendrían el mismo efecto en la renta disponible de los recreacionistas, no parece que tengan el mismo efecto en su utilidad. Creemos que estas diferencias pueden ser explicadas por la percepción por parte de los recreacionistas de que tienen derecho de acceder libremente al bosque y por un comportamiento estratégico de algunos encuestados que tratan de evitar el establecimiento de entradas para acceder a zonas actuales de libre acceso.

En lo concerniente al estudio sobre la conservación del lince Ibérico en España, hemos analizado el rol adoptado por los diferentes entrevistados cuando se enfrentan a un escenario de pago para implementar un programa de conservación. Al tratarse de un bien público, no todos los individuos actúan como consumidores, sino que pueden juzgar esta cuestión desde el punto de vista de las preferencias sociales. En nuestro estudio, ampliamos esta dicotomía para incorporar una nuevo tipo de individuo que destaca por su comportamiento estratégico. De hecho, este tipo de comportamiento es clave para entender las preferencias adoptadas. Como se esperaba, son los que están dispuestos a pagar una menor cantidad de dinero para preservar el lince ibérico. Asimismo, los resultados muestran que la mayoría de los encuestados no ven factible que se pueda poner en marcha el programa. Esto nos lleva a concluir que es relevante esforzarse por elaborar escenarios que puedan resultar creíbles para los encuestados porque, de lo contrario, podríamos encontrarnos con un problema de sesgo hipotético.

Por lo tanto, esta tesis muestra que el reto al que se enfrentan las técnicas de no mercado para valorar los servicios de los ecosistemas de no mercado va más allá del la aplicación del método en sí mismo. También incluye cómo integrar los valores de la disposición a pagar en un análisis extendido que busca incorporar el valor económico de estos servicios. Sería necesario explorar los supuestos adicionales sobre esta integración, así como escenarios alternativos que podrían parecerse más a los mercados potenciales de estos servicios.

Al margen de lo concluido hasta ahora, otra contribución relevante de la tesis es la estimación de la acumulación neta del capital forestal. En el modelo teórico anteriormente mencionado hemos mostrado que, en consonancia con la literatura anterior, el Hamitoniano valor descontado y el Producto Nacional Neto Verde pueden ser valorados únicamente con valores del año en curso, lo cual facilita enormemente el análisis. No obstante, este supuesto está alejado de la realidad, ya que, entre otros supuestos, asume que existe una única especie. Si extendemos este modelo para permitir que existan distintas especies que crecen de forma diferente, mostramos que ya no es posible utilizar valores del año en curso porque el Producto Nacional Neto Verde incluiría valores a futuro. Para ello, hemos aplicado una integral de Volterra, integral que hasta el momento no había sido empleada en ningún estudio de contabilidad nacional verde. De este modo, mostramos que es posible utilizar este tipo de herramientas también dentro de este enfoque.

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APPENDICES

APPENDIX A: NECESSARY CONDITIONS FOR THE BENCHMARK MODEL

From equations [3.1] and [3.6], we can rearrange the Social Planner's problem as:

$$\max_{i(t),h(t)} \int_0^\infty U\left[Y\left(K(t), h(t), P(t)\right) - \phi\left(i(t)\right) - \psi\left(h(t)\right), A\left(R(t)\right)\right] e^{-rt}\, dt \qquad [A.1]$$

subject to [3.2], [3.3] and [3.4]. The current-value Hamiltonian [3.7] can be defined as:

$$\begin{aligned} H_c(t) &= U\left[Y\left(K(t), h(t), P(t)\right) - \phi\left(i(t)\right) - \psi\left(h(t)\right), A\left(R(t)\right)\right] \\ &+ \lambda_1(t)\left[i(t) - \delta K(t)\right] + \lambda_2(t)\left[g\left(R(t)\right) - h(t)\right] \qquad [A.2] \\ &+ \lambda_3(t)\left[e\left(K(t)\right) - \eta\left(g\left(R(t)\right) - h(t)\right) - \sigma P(t)\right] \end{aligned}$$

The Pontryagin's maximum principle provides the necessary conditions that must be fulfilled by the optimal control problem. On the one hand, there is the requirement that the current-value Hamiltonian be maximized with respect to the control variables at every point of time such that:

$$\frac{\partial H_c(t)}{\partial u(t)} = 0$$

where *u* is the control variable. Thus, the necessary conditions for our benchmark model are calculated as follows:

- Control variable *i(t)*:

$$\frac{\partial H_c(t)}{\partial i(t)} = -\frac{\partial U(t)}{\partial C(t)} \frac{\partial \phi\left(i(t)\right)}{\partial i(t)} + \lambda_1(t) = 0$$

where we can obtain $\lambda_1(t)$:

$$\lambda_1(t) = \frac{\partial \phi\left(i(t)\right)}{\partial i(t)} p(t) = \phi'\left(i(t)\right) p(t) = p_i(t) \qquad [A.3]$$

- Control variable *h(t)*:

$$\frac{\partial H_c(t)}{\partial h(t)} = \left[\frac{\partial U(t)}{\partial C(t)}\frac{\partial Y(K(t),h(t),P(t))}{\partial h(t)} - \frac{\partial U(t)}{\partial C(t)}\frac{\partial \psi(h(t))}{\partial h(t)} - \lambda_2(t) + \lambda_3(t)\eta\right] = 0$$

Following the same procedure as above, $\lambda_2(t)$ is:

$$\lambda_2(t) = \frac{\partial Y(K(t),h(t),P(t))}{\partial h(t)} p(t) - \frac{\partial \psi(h(t))}{\partial h(t)} p(t) + \lambda_3(t)\eta \qquad \text{[A.4]}$$

Defining (to simplify)

$$\frac{\partial Y(K(t),h(t),P(t))}{\partial h(t)} p(t) = p_w(t) \text{ and } \frac{\partial \psi(h(t))}{\partial h(t)} p(t) = p_h(t)$$

[A.4] is now:

$$\lambda_2(t) = p_w(t) - p_h(t) + \lambda_3(t)\eta \qquad \text{[A.5]}$$

On the other hand, the maximum principle also requires the following necessary condition to be met:

$$\dot{\lambda}(t) = -\frac{\partial H_c(t)}{\partial y(t)} + r\lambda(t) \qquad \text{[Equation of motion for } \lambda(t)\text{]}$$

where *y* is the state variable of the model. Hence, the only condition needed to solve our problem is:

$$\dot{\lambda}_1(t) = -\frac{\partial H_c(t)}{\partial K(t)} + r\lambda_1(t) \qquad \text{[Equation of motion for } \lambda_1(t)\text{]} \qquad \text{[A.6]}$$

Noting [A.3] and assuming that prices are constant over time, [A.6] is equal to 0, so:

$$\dot{\lambda}_1(t) = -\left[\frac{\partial U(t)}{\partial C(t)}\frac{\partial Y(K(t),h(t),P(t))}{\partial K(t)} - \lambda_1(t)\delta + \lambda_3(t)\frac{\partial e(K(t))}{\partial K(t)}\right] + r\lambda_1(t) = 0 \qquad \text{[A.7]}$$

From this equation, we can derive $\lambda_1(t)$ such that:

$$\lambda_1(t) = \frac{1}{(r+\delta)}\left[\frac{\partial Y(K(t),h(t),P(t))}{\partial K(t)}p(t) + \lambda_3(t)\frac{\partial e(K(t))}{\partial K(t)}\right] \quad \text{[A.8]}$$

Considering [A.3], it is easy to obtain $\lambda_3(t)$:

$$B_3(t) = \left[\frac{(r+\delta)\dfrac{\partial \phi(i(t))}{i(t)} - \dfrac{\partial Y(K(t),h(t),P(t))}{\partial K(t)}}{\dfrac{\partial e(K(t))}{\partial K(t)}}\right]p(t) \quad \text{[A.9]}$$

Using equation [A.5] we can directly obtain $\lambda_2(t)$:

$$\lambda_2(t) = p_w(t) - p_h(t) + B_1(t)\eta \quad \text{[A.10]}$$

where $B_1(t)$ is defined as in [3.16].

In order to estimate the 'linearized' current-value Hamiltonian and GNNP, we can rewrite [A.2] as:

$$\begin{aligned} H_c(t) &= S[Y(K(t),h(t),P(t)) - \phi(i(t)) - \psi(h(t)), A(R(t))] \\ &+ [Y(K(t),h(t),P(t)) - \phi(i(t)) - \psi(h(t))]\frac{\partial U(t)}{\partial[Y(K(t),h(t),P(t)) - \phi(i(t)) - \psi(h(t))]} \\ &+ \frac{\partial U(t)}{\partial A(R(t))}A(R(t)) + \lambda_1(t)[i(t) - \delta K(t)] + \lambda_2(t)[g(R(t)) - h(t)] \\ &+ \lambda_3(t)[e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t)] \end{aligned} \quad \text{[A.11]}$$

Taking out consumer surplus, [A.11] is

$$\begin{aligned} \hat{H}_c(t) &= [Y(K(t),h(t),P(t)) - \phi(i(t)) - \psi(h(t))]\frac{\partial U(t)}{\partial[Y(K(t),h(t),P(t)) - \phi(i(t)) - \psi(h(t))]} \\ &+ \frac{\partial U(t)}{\partial A(R(t))}A(R(t)) + \lambda_1(t)[i(t) - \delta K(t)] + \lambda_2(t)[g(R(t)) - h(t)] \\ &+ \lambda_3(t)[e(K(t)) - \eta(g(R(t)) - h(t)) - \sigma P(t)] \end{aligned} \quad \text{[A.12]}$$

Rearranging and replacing $\lambda_1(t)$, $\lambda_2(t)$ and $\lambda_3(t)$ by [3.10], [3.11] and [3.12], we obtain [3.18].

APPENDIX B. NECESSARY CONDITIONS FOR THE MODEL INVOLVING A SINGLE SPECIES AND A CONSTANT DECAY EXPONENTIAL GROWTH FUNCTION

We follow the same procedure as that for the benchmark model, albeit with a few modifications. From equations [3.22] and [3.24], we can rearrange the Social Planner's problem as:

$$\max_{i(t),x(t)} \int_0^{\infty} U\left[Y(K(t),x(t),P(t)) - \phi(i(t)) - \varsigma(x(t)), A(R(t))\right] e^{-rt}\,dt \quad \text{[B.1]}$$

subject to [3.2], [3.25], [3.26] and [3.27]. The current-value Hamiltonian is:

$$\begin{aligned} H_c(t) &= U\left[Y(K(t),x(t),P(t)) - \phi(i(t)) - \varsigma(x(t)), A(R(t))\right] + \lambda_1(t)\left[i(t) - \delta K(t)\right] \\ &+ \lambda_2(t)G(t) + \lambda_3(t)\left[e(K(t)) - \eta G(t) - \sigma P(t)\right] + \lambda_4(t)\left[x(t) - bG(t)\right] \end{aligned} \quad \text{[B.2]}$$

The necessary conditions are:

- Control variable *i(t)*:

  The result is the same as in the benchmark model, i.e. [A.3].

- Control variable *x(t)*:

$$\frac{\partial H_c(t)}{\partial x(t)} = \left[\frac{\partial U(t)}{\partial C(t)}\frac{\partial Y(K(t),x(t),P(t))}{\partial x(t)} - \frac{\partial U(t)}{\partial C(t)}\frac{\partial \varsigma(x(t))}{\partial x(t)} + \lambda_4(t)\right] = 0 \quad \text{[B.3]}$$

where we can now easily obtain $\lambda_4(t)$:

$$\lambda_4(t) = \left[\frac{\partial Y(K(t),x(t),P(t))}{\partial x(t)} p(t) - \frac{\partial \varsigma(x(t))}{\partial x(t)} p(t)\right] \quad \text{[B.4]}$$

Defining (to simplify)

$$\frac{\partial Y(K(t),x(t),P(t))}{\partial x(t)}p(t)=p_{F_x} \text{ and } \frac{\partial \varsigma(x(t))}{\partial x(t)}p(t)=p_x(t) \qquad \text{[B.5]}$$

[B.4] is now:

$$\lambda_4(t)=p_{Fx}(t)-p_x(t) \qquad \text{[B.6]}$$

Furthermore, the equations of motion needed to solve the problem are:

$$\dot{\lambda}_1(t)=-\frac{\partial H_c(t)}{\partial K(t)}+r\lambda_1(t) \qquad \text{[Equation of motion for } \lambda_1(t)\text{]}$$

where

$$\dot{\lambda}_1(t)=-\left[\frac{\partial U(t)}{\partial C(t)}\frac{\partial Y(K(t),x(t),P(t))}{\partial K(t)}-\lambda_1(t)\delta+\lambda_3(t)\frac{\partial e(K(t))}{\partial K(t)}\right]+r\lambda_1(t) \qquad \text{[B.7]}$$

Considering equation [A.3], [B.7] is equal to 0, so we can obtain $\lambda_1(t)$:

$$\lambda_1(t)=\frac{1}{(r+\delta)}\left[\frac{\partial Y(K(t),x(t),P(t))}{\partial K(t)}p(t)+\lambda_3(t)\frac{\partial e(K(t))}{\partial K(t)}\right] \qquad \text{[B.8]}$$

Using equation [A.3], we can directly derive $\lambda_1(t)$:

$$\lambda_3(t)=\left[\frac{(r+\delta)\frac{\partial \phi(i(t))}{\partial i(t)}-\frac{\partial Y(K(t),x(t),P(t))}{\partial K(t)}}{\frac{\partial e(K(t))}{\partial K(t)}}\right]p(t) \qquad \text{[B.9]}$$

The following equation of motion needed is:

$$\dot{\lambda}_4(t)=-\frac{\partial H_c(t)}{\partial G(t)}+r\lambda_4(t) \qquad \text{[Equation of motion for } \lambda_4(t)\text{]}$$

where

$$\dot{\lambda}_4(t)=-\lambda_2(t)+\lambda_3(t)\eta+\lambda_4(t)[b+r] \qquad \text{[B.10]}$$

Considering equation [B.6]:

$$\dot{\lambda}_4(t)=-\lambda_2(t)+\lambda_3(t)\eta+(p_{Fx}(t)-p_x(t))[b+r] \qquad \text{[B.11]}$$

$\dot{\lambda}_4(t)=0$ because prices are assumed to be constant over time, so we can obtain $\lambda_2(t)$:

$$\lambda_2(t)=\lambda_3(t)\eta+\left(p_{F_x}(t)-p_x(t)\right)[b+r] \qquad \text{[B.12]}$$

Substituting [B.9] into [B.12]:

$$\lambda_2(t)=B_2(t)\eta+\left(p_{F_x}(t)-p_x(t)\right)[b+r] \qquad \text{[B.13]}$$

where $B_2(t)$ is defined as in [3.34].

In order to estimate the 'linearized' current-value Hamiltonian and GNNP, we can re-write [B.2] as:

$$\begin{aligned} H_c(t) &= S[Y(K(t),x(t),P(t))-\phi(i(t))-\varsigma(x(t)),A(R(t))] \\ &+[Y(K(t),x(t),P(t))-\phi(i(t))-\varsigma(x(t))]\frac{\partial U(t)}{\partial[Y(K(t),x(t),P(t))-\phi(i(t))-\varsigma(x(t))]} \\ &+A(R(t))\frac{\partial U(t)}{\partial A(R(t))}+\lambda_1(t)[i(t)-\delta K(t)]+\lambda_2(t)G(t) \\ &+\lambda_3(t)[e(K(t))-\eta G(t)-\sigma P(t)]+\lambda_4(t)[x(t)-bG(t)] \end{aligned} \qquad \text{[B.14]}$$

Taking out consumer surplus, replacing $\lambda_1(t), \lambda_2(t)$, $\lambda_3(t)$ and $\lambda_4(t)$ by [3.10], [3.29], [3.30] and [3.31], respectively, and rearranging, we obtain [3.35].

APPENDIX C: NECESSARY CONDITIONS FOR THE MODEL INVOLVING DIFFERENT TYPES OF SPECIES

As explained in the text, in this case we are using the Hamiltonian, rather than the current-value Hamiltonian because it greatly facilitates the derivation of the necessary conditions. Note that we now have a state equation defined as a Volterra integral equation, so these conditions must differently be calculated (Vinokurov, 1969; Kamien and Schwartz, 1991).

From equations [3.39] and [3.41], we can rearrange the Social Planner's problem as:

$$\max_{i(t),x(t),b(t)} \int_0^\infty U\left[Y\left(K(t), x(t), b(t), P(t)\right) - \phi(i(t)) - \varsigma(x(t)), A(R(t))\right] e^{-rt} dt \qquad \text{[C.1]}$$

subject to [3.2], [3.42], [3.43] and [3.44]. The Hamiltonian is now:

$$H(t) = U\left[Y\left(K(t), x(t), b(t), P(t)\right) - \phi(i(t)) - \varsigma(x(t)), A(R(t))\right] e^{-rt}$$

$$+ \lambda_1(t)\left[i(t) - \delta K(t)\right] + \lambda_2(t) G(t) + \lambda_3(t)\left[e(K(t)) - \eta G(t) - \sigma P(t)\right] \qquad \text{[C.2]}$$

$$+ \lambda_4(t) x(t) - \int_t^\infty \lambda_4(s) b(t) x(t) e^{-b(t)(s-t)} ds$$

From [C.2], the necessary conditions are:

- Control variable *i(t)*:

$$\frac{\partial H(t)}{\partial i(t)} = -\frac{\partial U(t)}{\partial C(t)} \frac{\partial \phi(i(t))}{\partial i(t)} e^{-rt} + \lambda_1(t) = 0$$

where we can now obtain $\lambda_1(t)$:

$$\lambda_1(t) = \frac{\partial \phi(i(t))}{\partial i(t)} p(t) e^{-rt} = \phi^{'}(i(t)) p(t) e^{-rt} = p_i(t) e^{-rt} \qquad \text{[C.3]}$$

- Control variable *x(t)*:

$$\frac{\partial H(t)}{\partial x(t)} = \begin{bmatrix} \frac{\partial U(t)}{\partial C(t)} \frac{\partial Y(K(t), x(t), b(t), P(t))}{\partial x(t)} e^{-rt} - \frac{\partial U(t)}{\partial C(t)} \frac{\partial \varsigma(x(t))}{\partial x(t)} e^{-rt} \\ + \lambda_4(t) - b(t) \int_t^{\infty} \lambda_4(s) e^{-b(t)(s-t)} ds \end{bmatrix} = 0 \qquad [C.4]$$

Defining (to simplify)

$$\frac{\partial Y(K(t), x(t), b(t), P(t))}{\partial x(t)} p(t) = p_{F_b}(t) \qquad [C.5]$$

and considering [C.4]:

$$\lambda_4(t) - b(t) \int_t^{\infty} \lambda_4(s) e^{-b(t)(s-t)} ds = -p_{Fb}(t) e^{-rt} + p_x(t) e^{-rt} \qquad [C.6]$$

- Control variable *b(t)*:

$$\frac{\partial H(t)}{\partial b(t)} = \left[\frac{\partial U(t)}{\partial C(t)} \frac{\partial Y(K(t), x(t), b(t), P(t))}{\partial b(t)} e^{-rt} - x(t) \int_t^{\infty} \lambda_4(s) e^{-b(t)(s-t)} ds \right] = 0 \qquad [C.7]$$

Defining (to simplify)

$$\frac{\partial Y(K(t), x(t), b(t), P(t))}{\partial b(t)} p(t) = p_b(t) \qquad [C.8]$$

[C.7] can be rewritten as:

$$\int_t^{\infty} \lambda_4(s) e^{-b(t)(s-t)} ds = \frac{p_b(t) e^{-rt}}{x(t)} \qquad [C.9]$$

Substituting [C.9] into [C.6], we can obtain $\lambda_4(t)$:

$$\lambda_4(t) = e^{-rt} \left[p_x(t) - p_{Fb}(t) + b(t) \frac{p_b(t)}{x(t)} \right] \qquad [C.10]$$

Furthermore, the maximum principle also requires the following conditions to be met:

$$\lambda_1(t) = \frac{\partial H(t)}{\partial K(t)}$$

where

$$\lambda_1(t)=\frac{1}{1+\delta}\left[\frac{\partial Y(K(t),x(t),b(t),P(t))}{\partial K(t)}p(t)e^{-rt}+\lambda_3(t)\frac{\partial e(K(t))}{\partial K(t)}\right] \quad [C.11]$$

and

$$\lambda_4(t)=\frac{\partial H(t)}{\partial G(t)}$$

where

$$\lambda_4(t)=\lambda_2(t)-\lambda_3(t)\eta \quad [C.12]$$

From [C.11] and taking into account [C.3], we can obtain $\lambda_3(t)$:

$$\lambda_3(t)=\left[\frac{(1+\delta)\frac{\partial\phi(i(t))}{\partial i(t)}-\frac{\partial Y(K(t),x(t),b(t),P(t))}{\partial K(t)}}{\frac{\partial e(K(t))}{\partial K(t)}}p(t)\right]e^{-rt} \quad [C.13]$$

Using [C.10] and [C.13]:

$$\lambda_2(t)=\left[\begin{array}{l} p_x(t)-p_{Fb}(t)+b(t)\frac{p_b(t)}{b(t)} \\ +\frac{(1+\delta)\frac{\partial\phi(i(t))}{\partial i(t)}-\frac{\partial Y(K(t),x(t),b(t),P(t))}{\partial K(t)}}{\frac{\partial e(K(t))}{\partial K(t)}}p(t)\end{array}\right]e^{-rt} \quad [C.14]$$

where $B_3(t)$ is defined as in [3.52].

In order to estimate the Hamiltonian and GNNP, we can rewrite [C.2] as:

$$\begin{aligned} H(t)=\; & S[Y(K(t),x(t),b(t),P(t))-\phi(i(t))-\varsigma(x(t)),A(R(t))]e^{-rt} \\ & +[Y(K(t),x(t),b(t),P(t))-\phi(i(t))-\varsigma(x(t))]\frac{\partial U(t)}{\partial[Y(K(t),x(t),b(t),P(t))-\phi(i(t))-\varsigma(x(t))]}e^{-rt} \\ & +\frac{\partial U(t)}{\partial A(R(t))}A(R(t))e^{-rt}+\lambda_1(t)[i(t)-\delta K(t)]+\lambda_2(t)G(t) \\ & +\lambda_3(t)[e(K(t))-\eta G(t)-\sigma P(t)]+\lambda_4(t)x(t)-\int_t^{\infty}\lambda_4(s)b(t)x(t)e^{-b(t)(s-t)}ds \end{aligned} \quad [C.15]$$

Taking out consumer surplus, replacing $\lambda_1(t),\lambda_2(t)$, $\lambda_3(t)$ and $\lambda_4(t)$ by [3.46], [3.47], [3.48] and [3.49], respectively, and rearranging, we obtain [3.53].

APPENDIX D. MODEL INVOLVING A SINGLE SPECIES, A CONSTANT DECAY EXPONENTIAL GROWTH FUNCTON AND FELLING

As explained in the text, the optimal control model developed in section 3.2.1 may also be developed if trees are felled. However, the mathematical complexity is unavoidable.

Let us now suppose that after *T* years (exogenous) all trees are taken out, so they do not grow anymore. This means that the accumulated growth in wood of each tree at the time of felling is defined by:

$$M(t,b,s) = M(T,b) = \int_0^T e^{-bs} ds = M$$ [D.1]

because we still assume that $G_0 = 1$. As *T* is exogenous, *M* is a constant parameter determined by the growth function. The growth of all vintages at time t, *G(t)*, is therefore given by:

$$G(t) = \int_{t-T}^{t} x(s) e^{-b(t-s)} ds \qquad [D.2]$$

and the function that accumulates all the past growth, *R(t)*, is:

$$R(t) = \int_{t-T}^{t} G(z) dz = \int_{t-T}^{t} \left(\int_{t-T}^{z} x(s) e^{-b(z-s)} ds \right) dz \qquad [D.3]$$

The Social Planner's problem is [3.22] subject to [3.2], [D.2], [D.3], the appropriate transversality conditions, and a modified version of [3.4]:

$$\dot{P}(t) = e(K(t)) - \eta \dot{R}(t) - \sigma P(t) \qquad [D.4]$$

The economy's total production at time *t* is defined as:

$$Y(K(t), x(t-T), P(t)) = C(t) + \phi(i(t)) + \varsigma(x(t)) \qquad [D.5]$$

We can take the time derivatives of [D.2] and [D.3]:

$$\dot{G}(t) = x(t) - x(t-T)e^{-bT} - b\int_{t-T}^{t} x(s)e^{bs-bt}ds$$
$$= x(t) - x(t-T)e^{-bT} - b\int_{t-T}^{t} x(s)e^{-b(t-s)}ds = x(t) - x(t-T)e^{-bT} - bG(t) \quad \text{[D.6]}$$

where the three terms of this expression are: (i) the growth added by the additional plantations (which add one unit of growth because $G_0 = 1$), (ii) the growth 'taken out' by felling the trees that reached their maturity (after *T* years), and (iii) the decay in the growth of all the active vintages because they are now one period older.

And:

$$\dot{R}(t) = \int_{t-T}^{t} x(s)e^{bs-bt}ds - x(t-T)\int_{t-T}^{t} e^{-b(T-t+z)}dz$$
$$= \int_{0}^{t} x(s)e^{-b(t-s)}ds - x(t-T)\int_{t-T}^{t} e^{-b(T-t+z)}dz = G(t) - x(t-T)\int_{t-T}^{t} e^{-b(T-t+z)}dz \quad \text{[D.7]}$$

Considering [D.1], [D.7] can be rewritten as:

$$\dot{R}(t) = G(t) - x(t-T)M \quad \text{[D.8]}$$

because

$$\int_{t-T}^{t} e^{-b(T-t+z)}dz = \int_{0}^{T} e^{-bz}dz = M$$

so [D.4] can now be defined as:

$$\dot{P}(t) = e(K(t)) - \eta[G(t) - x(t-T)M] - \sigma P(t) \quad \text{[D.9]}$$

The Social Planner's problem is no longer a conventional optimal control model, as there are variables at time *t* and time-delayed variables. This changes the way of solving the problem and the current-value Hamiltonian, as we now have a time-delayed Hamiltonian (Frankena, 1975; Kamien and Schwartz, 1991), so we cannot only use current values (see Caparrós (2009) for an application to carbon sequestration). We thus confirm that making a minimum change, such as the one shown here, the conventional model begins to be modified and becomes more complex. Only if there had not been the possibility of felling, the optimal control model would have remained unchanged.

We can rewrite the Social Planner's problem as before, while replacing [D.2] by [D.6] and [D.3] by [D.8]. The current-value Hamiltonian of this problem is:

$$
\begin{aligned}
H_c(t) &= U[C(t), A(R(t))] + \lambda_1(t)[i(t) - \delta K(t)] + \lambda_2(t)[G(t) - x(t-T)M] \\
&+ \lambda_3(t)[e(K(t)) - \eta(G(t) - x(t-T)M) - \sigma P(t)] \\
&+ \lambda_4(t)[x(t) - x(t-T)e^{-bT} - bG(t)]
\end{aligned} \qquad [D.10]
$$

As shown, [D.10] depends now on two points in time.

Unlike previous cases, necessary conditions for an optimal control model involving a time-delayed response are different (Frankena, 1975; Kamien and Schwartz, 1991):

$$\dot{\lambda}(t) = -\frac{\partial H_c(t)}{\partial y(t)} - \left|\frac{\partial H_c(t)}{\partial y(t-T)}\right|_{(t+T)} + r\lambda(t) \qquad [t_0 \le t \le t_1 \quad \tau] \qquad [D.11]$$

$$\frac{\partial H_c(t)}{\partial u(t)} + \left|\frac{\partial H_c(t)}{\partial u(t-T)}\right|_{(t+T)} = 0 \qquad [t_0 \le t \le t_1 \quad \tau] \qquad [D.12]$$

$$\dot{\lambda}(t) = -\frac{\partial H_c(t)}{\partial y(t)} + r\lambda(t) \qquad [t_1 - \tau \le t \le t_1] \qquad [D.13]$$

$$\frac{\partial H_c(t)}{\partial u(t)} = 0 \qquad [t_1 - \tau \le t \le t_1] \qquad [D.14]$$

where y is the state variable and u the control variable. By adapting these conditions to our problem, $t_0 = 0$, $t_1 = \infty$ and $\tau = T$, so we would only need to use [D.11] and [D.12].

Nevertheless, the derivation of this problem is so complex that we found no reasonable expressions for $\lambda_1(t)$, $\lambda_2(t)$, $\lambda_3(t)$ and $\lambda_4(t)$. In any case, it is to be expected that if we would apply the two necessary conditions indicated above, GNNP would also be defined in two points in time.

APPENDIX E. NET ACCUMULATION OF NATURAL CAPITAL[115]

We noted above that the term $\left(p_w(t)-p_h(t)\right)\left[g(R(t))-h(t)\right]$ in [3.18] does not adequately reflect the change in the value of natural capital (forest biomass in our case). This is because the model assumes that there is a single species and does not take into account the age-class structure and assumes that there is a single species. As explained in the second chapter of this dissertation, Vincent (1999a) calculates net accumulation of natural capital as the physical variation in stocks multiplied by the shadow price expressed in terms of future returns to the natural capital stock. This price is therefore a proxy for the marginal 'net price'. He also analyzes the different existing proposals in order to include net investment of net accumulation in national accounts. It follows that from a forest where felling in done once at time *T* there are two methods: El Serafy variation[116] and net-price method (see chapter 2). Caparrós et al. (2003) propose an alternative approach that confirms the results of El Serafy variation method, which is a particular case when all trees are cut down upon reaching age *T*. Let us see this approach by following the simplification of this model made by Caparrós (2010).

Capital balances

The capital value of a forest, *W*, is:

$$W(t)=\int_t^{\infty} w(s)e^{-r(s-t)}ds \qquad [E.1]$$

[115] Note that some subindexes in this appendix differ from those used throughout the dissertation. They use facilitates the analysis.

[116] Net accumulation in the year of the felling is approximately equal to the current rent (with negative sign). The rest of years, there is positive net accumulation, as forests are approaching the time of felling.

where w is the capital income obtained by the forest owner (it may, for example, include capital income from timber, grazing, hunting, recreational services and carbon, among others). That is, capital reflects the discounted stream of future net benefits, where t is the valuation year. Estimating this value at the beginning and at the end of the period and calculating the difference is the same as multiplying the change in the physical stock of natural capital by its costate variable (Caparrós et al., 2003), which is expressed in terms of future returns to the stock, as in Vincent (1999b).

To make the explanation easier, let us assume that wood is the only source of profit. This definition is closely related to that given by the SNA: *"The value of a fixed asset to its owner at any point of time is determined by the present value of the future capital services (that is, the sum of the values of the stream of future rentals less operating costs discounted to the present period) that can be expected over its remaining service life. Consumption of fixed capital is measured by the decrease, between the beginning and the end of the current accounting period, in the present value of the remaining sequence of expected future benefits"* (ISWGNA, 2008: para. 6246).

Caparrós et al. (2003) apply the AAS method explained in the second chapter of this dissertation. According to this methodology, the capital balance of a forest (or an agroforestry system) is divided into two balances: (i) the production in progress (forest stands[117]) balance, *PP*, and (ii) the fixed capital balance (in our case only land as we assume that there is no man-made capital in the forestry sector), *L*. Hence, and taking out any source of time-dependence:

$$W = PP + L \qquad [E.2]$$

Fixed capital balance: Land value is defined as remnant:

$$L = \int_t^{\infty} we^{-r(s-t)}ds - PP \qquad [E.3]$$

Products in progress balance: As the only product in progress considered is standing timber, its value can be estimated by the stream of future net returns:

$$PP_t = \int_t^{T} (p_{ws} - p_{hs})q_{hs}e^{-r(s-t)}ds \qquad [E.4]$$

[117] That is, the standing timber at the present time.

where q_{hs} indicates the quantity of timber felled.

However, trees are often cut selectively, at least in most European forests, so we can only estimate the probability that a living tree of the diametric-age *d* (at which we would have to associate an age) is felled in the diametric-age *s*, where $j > d$ [118]. This would imply knowing future forestry for each of the species available, which may be complex if selective felling processes are completed. However, this probability may be determined on the basis of historical extraction data, thus determining the conditional probability, π_{jd}, that a living tree of the diametric-age *d* is felled at the diametric-age *j*:

$$\pi_{jd} = \Pr\left(\frac{j}{d}\right) = \frac{q_{hj}}{\sum_{t=Nd}^{n} q_{ht}} \qquad [E.5]$$

where N_d is the number of diameter-ages *d*, arranging all classes from small to large.

The price for not felled standing timber, p_p, is, for each diameter class, the mathematical expectation formed with the prices that the timber will have at the future time of felling, consequently discounted

$$p_{pd} = E(p_h) = \sum_{j=Nd}^{N} \left(p_{wj} - p_{hj}\right)\pi_{jd}(1+r)^{(s_t - s_d)} \qquad [E.6]$$

where *N* is the total number of diameter classes, arranged by age, s_j is the average age of class *j*; and s_d is the age of the present diameter class *d*.

Using the calculated not felled timber prices, the production in progress is valued at the beginning, PP_i, and at the end, PP_f, on the accounting period, with the following expression:

$$PP_t = \sum_{j=1}^{n} p_{pjt} q_{pjt} \qquad j = i,\ f \qquad [E.7]$$

where [E.7] aggregates the estimated values for each diameter-class (q_{pjt} is the not felled timber for each diameter-class).

118 As described above, the forestry analyzed by Vincent (1999b) when describing the El Serafy variation method is a particular case in which the probability of harvesting is zero until the moment trees reach the diametric-age associated with age *T* when the probability of felling is one.

Used products in progress, PP_u, i.e. the timber extracted from the forest, is valued at the present market price for the different age classes:

$$PP_u = \sum_{j=1}^{n} \left(p_{wj} - p_{hj}\right) q_{hj} \qquad \text{[E.8]}$$

Being q_{hj} annual commercial gross growth of the diameter-class j that has not been extracted, total stock entrance, PP_e, or equivalently natural gross commercial growth[119], is valued according to the expression:

$$PP_e = \sum_{j=1}^{n} g_{pj} \, p_{pj} \qquad \text{[E.9]}$$

This expression multiplies (for each age class) the growth which has occurred over the period by its price, considering whether new timber growth is felled in the same year or remains standing (assuming that extraction occurs at the end of the period).

According to the above, the revaluation of timber during the period is estimated as remainder:

$$PP_{rem} = PP_f - PP_i + PP_u - PP_e \qquad \text{[E.10]}$$

The main source of revaluations is the change in the value of standing timber when changing from the diameter-class d to the diameter-class ($d+1$), if the price of ($d+1$) is larger than that of d. That is, the growth occurred in the forest not only increases the existing biomass, but also the value of the existing timber at the beginning of the period. This effect can only occur when the timber price varies according to the diameter, or it is necessary to wait until the time of felling, as otherwise $PP_{rem} = 0$. This is not the case in the general equilibrium models described above, they conclude that it is sufficient to subtract extractions to gross growth of the year (valued by the single price of timber). In our benchmark model we showed that the following expression should be added to GNNP:

$$\left[\left(p_w - p_h\right)g(R) - \left(p_w - p_h\right)h\right] \qquad \text{[E.11]}$$

[119] Natural gross commercial growth equals total stock entrance as the only production in progress considered in this paper is timber.

The first term of [E.11] represents gross natural growth valued at the stumpage price (a single price for all diameter-classes) such that:

$$PP_e = (p_w - p_h)g(R) \quad \text{[E.12]}$$

$$PP_u = (p_w - p_h)h \quad \text{[E.13]}$$

because in this case

$$g = \sum_j g_{pj} \quad \text{[E.14]}$$

$$h = \sum_j q_{jj} \quad \text{[E.15]}$$

$$p_p = (p_w - p_h) = p_{pj} = (p_{wj} - p_{hj}) \quad \text{[E.16]}$$

It makes no sense to distinguish among diameter-classes if all of them have the same price and it is not necessary to wait for the time of felling. The difference between the final value and the initial value, is given in the methodology proposed by Caparrós et al. (2003) as:

$$PP_f - PP_i = PP_{rem} + PP_e - pp_U \quad \text{[E.17]}$$

If we assume that there is a single price of timber and that forests grow instantaneously, $PP_{rem} = 0$, so expression [E.17] is equivalent to the bracket shown in [E.11]. Vincent (1999a) already indicates that the result that the only change to be made in GNNP is to include [E.11] depends critically on the assumption of a single type of forest and a single price for standing timber (and we should also assume that forest grow instantaneously). For the more realistic case that there are various types of forests and that the price of timber depends only on the diameter he suggests directly working with discounted future returns ($PP_f - PP_i$). The proposal of Caparrós et al. (2003) is similar, albeit with the addition of specifications for different sources of change in the value between the initial time and the final time over the accounting period. Indeed, inn this chapter we have shown that, when there are different types of species we cannot obviate the future.

APPENDIX F: NECESSARY CONDITIONS FOR THE MODEL INVOLVING NON-MARKET GOODS AND SERVICES

From [3.54] and [3.55], we can rearrange the Social Planner's problem as:

$$\max_{i(t),h(t),\upsilon(t)} \int_t^{\infty} U\left[Y\left(K(t),h(t),P(t)\right)-\phi(i(t))-\psi(h(t))-\omega(\upsilon(t)),A(\upsilon(t),R(t))\right]e^{-rt}dt \qquad [F.1]$$

subject to [3.2], [3.3] and [3.4]. The current-value Hamiltonian is:

$$\begin{aligned}&H_c(t)=U\left[Y\left(K(t),h(t),P(t)\right)-\phi(i(t))-\psi(h(t))-\omega(\upsilon(t)),A(\upsilon(t),R(t))\right]\\&+\lambda_1(t)(i(t)-\delta K(t))+\lambda_2(t)\left[g(R(t))-h(t)\right]\\&+\lambda_3(t)\left[e(K(t))-\eta\left[g(R(t))-h(t)\right]-\sigma P(t)\right]\end{aligned} \qquad [F.2]$$

The only difference with respect to the necessary conditions obtained for the benchmark model is that there is a new control variable $\upsilon(t)$ so that a new condition for this variable is needed:

$$\frac{\partial H_c(t)}{\partial \upsilon(t)}=\left(-\frac{\partial U(t)}{\partial C(t)}\frac{\partial \omega(\upsilon(t))}{\partial \upsilon(t)}\right)+\frac{\partial U(t)}{\partial A(\upsilon(t),R(t))}\frac{\partial A(\upsilon(t),R(t))}{\partial \upsilon(t)}=0$$

$$\frac{\partial \omega(\upsilon(t))}{\partial \upsilon(t)}p(t)=\frac{\partial U(t)}{\partial A(\upsilon(t),R(t))}\frac{\partial A(\upsilon(t),R(t))}{\partial \upsilon(t)}=p_A(t) \qquad [F.3]$$

In order to estimate the Hamiltonian and GNNP, we can re-write [F.2] as:

$$
\begin{aligned}
H(t) &= S[Y(K(t),h(t),P(t)) - \phi(i(t)) - \beta(x(t)) - \omega(\upsilon(t)), A(R(t))]e^{-rt} \\
&+ [F(K(t),h(t),P(t)) - \phi(i(t)) - \beta(x(t)) - \omega(\upsilon(t))]\frac{\partial U(t)}{\partial Y(K(t),h(t),P(t)) - \phi(i(t)) - \beta(x(t)) - \omega(\upsilon(t)), A(R(t))} e^{-rt} \\
&+ A(R(t))\frac{\partial U(t)}{\partial A(R(t))} e^{-rt} + \lambda_1(t)(i(t) - \delta K(t)) + \lambda_2(t)[g(R(t) - h(t))] \\
&+ \lambda_3(t)[e(K(t)) - \eta[g(R(t) - h(t))] - \sigma P(t)]
\end{aligned}
\quad [F.4]
$$

Taking out consumer surplus, replacing $\lambda_1(t), \lambda_2(t)$ and $\lambda_3(t)$ by [3.10], [3.11] and [3.12], and rearranging, we obtain [3.60].

APPENDIX G. BOOKLET

THE IBERIAN LYNX

The Iberian Lynx (Pictures 9 and 10) is an endemic species from the Iberian Peninsula and, according to the International Union for Conservation of Nature (IUCN), it is the most endangered feline species on Earth together with the Bengali Tiger.

The census performed for this species in 1990 estimated that the population consisted of **1,100 specimens**.

The last census pointed out that current population did not exceed **200 specimens** (a **82%** fall in **15 years**).

Picture 9: *Iberian Lynx*

It is considered that population in 1990 (1,100 specimens) is the minimum threshold for the preservation of this species. If adequate measures are not put into place, this feline might disappear.

Although there are currently no captive breeding programs, results are still uncertain and do not imply the species' recovery in its natural habitat.

Picture 10: *Iberian Lynx*

The main threats for the Iberian Lynx are the lack of its basic prey (rabbit), the mortality caused by man, the destruction and alteration of the habitat and the isolation and fragmentation of their populations, as there are no ways of exchanging individuals.

FOREST RECREATIONAL USE

In what follows, we will ask you some questions about possible recreational visits to forests that you may make in the future. Take into account the potential visits we present:

These visits may solely be made to **stone pine** forests (previously described) or **cork oak** forests.

Cork oak is a native species that shares the same ecosystem than the Stone pine. In fact, these species are mixed in the southwest and northwest of Spain. Cork oak forests are also located in Extremadura. Flora and fauna are also abundant in cork oak forests.

Cork oak

These visits may be made to recreational areas and/or trails in which there are (or not) **infrastructures** (**tables**, **benches**, **toilets** and **swings**).

These visits may be made to recreational areas and/or trails in which there is the possibility of **picking mushrooms** (for example, saffron milk cap)

These visits may be made to recreational areas and/or trails in which there is the possibility of seeing **domesticated animals** (cows, horses, sheeps…).

Moreover, your visit would include a total expenditure derived from **GASOLINE COSTS** and from the hypothetical **ENTRANCE FEE** that the owner of the natural area would set for the Access.

Please, indicate to the interviewer that you have already finished and answer the question he/she will ask you. Thank you.

APPENDIX H: ORIGINAL SURVEY

GOBIERNO DE ESPAÑA | MINISTERIO DE EDUCACIÓN Y CIENCIA

INIA Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria

CSIC

Universidad Complutense Madrid

THIS SURVEY IS CONDUCTED WITHIN THE FRAMEWORK OF A RESEARCH PROJECT BY THE SPANISH NATIONAL RESEARCH COUNCIL, THE NATIONAL INSTITUTE FOR AGRICULTURAL AND FOOD RESEARCH AND TECHNOLOGY AND THE COMPLUTENSE UNIVERSITY OF MADRID.

WE ARE VERY GRATEFUL FOR YOUR COOPERATION. THIS SURVEY IS COMPLETELY ANONYMOUS. KEEP THEFOLLOWING INFORMATION IN MIND WHEN IT COMES TO ANSWERING:

1) **THERE ARE NEITHER GOOD NOR BAD ANSWERS. WE ONLY WANT TO KNOW YOUR OPINION.**
2) **TRY TO ANSWER ALL THE QUESTIONS OF THE SURVEY. SOMETIMES, THE LACK OF A CONCRETEDATA SERIOUSLY DAMAGES THE UTILITY OF THE SURVEY.**
3) **TAKE THE TIME TO DO IT.**

OUR INTERVIEWER WILL ASK YOU THE QUESTIONS AND GUIDEYOU IN FILLING OUT THE SURVEY.

Survey No.: **Type: 1**

Please indicate your level of agreement/ disagreement with the following statements (1 means ‘Completely disagreement ’ and 5 ‘Full agreement’)

1. The conservation of the Spanish forests is important for me because they are places I like to visit with recreational purposes.

1 2 3 4 5
Complete disagreement Full agreement

2. The conservation of the native forests is important for me because it gives us the possibility of seeing beautiful landscapes.

1 2 3 4 5
Complete disagreement Full agreement

3. I would like to know that the conservation of the Spanish forests will be retained in its current state, even if I knew that I and my family would never visit them with recreational purposes.

1 2 3 4 5
Complete disagreement Full agreement

4. Although I have never thought about traveling there, I think that we must do our utmost to halt deforestation in the Amazon.

1 2 3 4 5
Complete disagreement Full agreement

5. Forest fires must be avoided at all costs, as the landscape that remains after a fire is very ugly.

1 2 3 4 5
Complete disagreement Full agreement

6. Could you indicate to us which of the following forests you like best? (MARK JUST ONE):

- ☐ STONE PINE ☐ SCOTS PINE
- ☐ HOLM OAK ☐ EUCALYPTUS
- ☐ CORK OAK ☐ EUROPEAN BEECH
- ☐ I do not distinguish between one forest and another
- ☐ Other (specify):

7. Did you know that tree plantations (reforestations) have been carried out over the last 15 years in Spain by means of subsidies granted by the Spanish public administration and by the European Union?

☐ Yes ☐ No

8. Do you know where the main masses of stone pine forests are located in Spain?

☐ Yes ☐ No

9. Could you tell us what your opinion is in regards to the implementation of a reforestation with STONE PINE in the southwest of Spain?

1 2 3 4 5
Strongly negative Very positive

Please read the booklet that we will provide to you as follows carefully in order to be able to answer the following questions with more information about the Iberian Lynx.

SHOW BOOKLET (Iberian Lynx)

The following questions will vary depending on the type of survey chosen[120]

Type 1

15. Suppose that a conservation fund to ensure that the Iberian Lynx population is back to the 1990 levels (1,100 specimens) is established. Consider that it is the minimum threshold necessary to ensure its preservation. This fund would be managed by a NGO which would be subject to government auditing. The money would only be allocated for the described purpose.

Would you be willing to contribute 10 EUROS to this fund (this year alone)?

☐ Yes (*question16*) ☐ No (*question 17*)

16. (If you answered **YES** to the question15). Would you be willing to contribute **30 EUROS**?

☐ Yes ☐ No

17. (If you answered **NO** to the question 15). Would you be willing to contribute **5 EUROS**?

☐ Yes ☐ No

18. What would be the **MAXIMUM AMOUNT** of money you would be willing to contribute to the fund?

__________ **EUROS** (*question 23 if zero*)

☐ N/A (*question 23 if the answer to questions 15 and 17 were also NO*)

19. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** (adequate from society's point of view) to guarantee the preservation of the Iberian Lynx.
b) I pay because the preservation of the Iberian Lynx is important for **ME**.

20. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I paid the **REST OF SPANISH CITIZENS** (contributors) would pay the same quantity.
b) I thought that if I paid **REST OF SPANISH CITIZENS** (contributors) would pay, but not necessarily the same quantity.
c) I thought that if I paid even a small quantity, this could be **COMPENSATED** by the **PAYMENT OF OTHER CONTRIBUTORS**.
d) **I DID NOT** take into account the quantity that others would pay.

21. In short, which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** that **ALL SPANISH CITIZENS** (contributors) **PAY** to guarantee the preservation of the Iberian Lynx.
b) I pay because preserving the Iberian Lynx is **SOCIALLY CORRECT** and it justifies this **EFFORT ON MY PART**, even if others pay less.
c) I pay because preserving the Iberian Lynx is important for **ME** and it justifies this **EFFORT ON MY PART**, even if others pay less.
d) None of the above, explain your reasons:

22. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would pay **MORE**; please, indicate the quantity: ………………..
b) I would pay **LESS**; please, indicate the quantity: …………………
c) I would pay **THE SAME**.

(*Go to question 26*)

[120] Questions from 10 to 14 were designed to be used as a choice experiment for the implementation of reforestation processes in the southwest of Spain. These questions therefore correspond to a different study and are not reported here.

[*Questions for those who did not accept paying any amount of money in **questions 15-18***]

23. Could you tell us the reason why you did not accept paying any amount of money to contribute to the conservation fund for the Iberian Lynx?:

24. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would pay.
b) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would not pay either.
c) **I DID NOT** take into account the quantity that others would pay.

25. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would continue without paying.
b) I **WOULD PAY**; please, indicate the quantity: …………………

26. On a scale of 1 to 5, you found the content of **question 15** (*show it again*):

1 2 3 4 5
It is very unclear — Very clear

27. On a scale of 1 to 5, do you think it is possible to implement the proposal of establishing a fund introduced in **question 15**?

1 2 3 4 5
Impossible — Very possible

28. On a scale of 1 to 5, to what degree of security did you answer **question 15**?

1 2 3 4 5
I very much doubted — I am very sure

Type 2

15. Suppose that a conservation fund to ensure that the Iberian Lynx population is back to the 1990 levels (1,100 specimens) is established. Consider that it is the minimum threshold necessary to ensure its preservation. This fund would be managed by a NGO which would be subject to government auditing. The money would only be allocated for the described purpose.

If the contributions obtained proved insufficient to finance the project, funds would be reimbursed to those who paid. If more funds than were needed were collected, surplus would also be reimbursed to those who paid in proportion to their contribution.

Would you be willing to contribute **30 EUROS** to this fund (**this year alone**)?

☐ Yes (*question16*) ☐ No (*question 17*)

16. (If you answered **YES** to the question15). Would you be willing to contribute **45 EUROS**?

☐ Yes ☐ No

17. (If you answered **NO** to the question 15). Would you be willing to contribute **15 EUROS**?

☐ Yes ☐ No

18. What would be the **MAXIMUM AMOUNT** of money you would be willing to contribute to the fund?

____________ **EUROS** (*question 23 if zero*)

☐ N/A (*question 23 if the answer to questions 15 and 17 were also NO*)

19. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** (adequate from society's point of view) to guarantee the preservation of the Iberian Lynx.
b) I pay because the preservation of the Iberian Lynx is important for **ME**.

20. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I paid the **REST OF SPANISH CITIZENS** (contributors) would pay the same quantity.
b) I thought that if I paid **REST OF SPANISH CITIZENS** (contributors) would pay, but not necessarily the same quantity.
c) I thought that if I paid even a small quantity, this could be **COMPENSATED** by the **PAYMENT OF OTHER CONTRIBUTORS**.
d) **I DID NOT** take into account the quantity that others would pay.

21. In short, which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** that **ALL SPANISH CITIZENS** (contributors) **PAY** to guarantee the preservation of the Iberian Lynx.
b) I pay because preserving the Iberian Lynx is **SOCIALLY CORRECT** and it justifies this **EFFORT ON MY PART**, even if others pay less.
c) I pay because preserving the Iberian Lynx is important for **ME** and it justifies this **EFFORT ON MY PART**, even if others pay less.
d) None of the above, explain your reasons:

22. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would pay **MORE**; please, indicate the quantity: ………………..
b) I would pay **LESS**; please, indicate the quantity: …………………
c) I would pay **THE SAME**.

(*Go to question 26*)

[*Questions for those who did not accept paying any amount of money in* ***questions 15-18***]

23. Could you tell us the reason why you did not accept paying any amount of money to contribute to the conservation fund for the Iberian Lynx?:

24. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would pay.
b) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would not pay either.
c) **I DID NOT** take into account the quantity that others would pay.

25. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would continue without paying.
b) I **WOULD PAY**; please, indicate the quantity: …………………

26. On a scale of 1 to 5, you found the content of **question 15** (*show it again*):

1 2 3 4 5
It is very unclear Very clear

27. On a scale of 1 to 5, do you think it is possible to implement the proposal of establishing a fund introduced in **question 15**?

1 2 3 4 5
Impossible Very possible

28. On a scale of 1 to 5, to what degree of security did you answer **question 15**?

1 2 3 4 5
I very much doubted I am very sure

Type 3

15. Suppose that the government proposes an income tax increase in order to finance a project aimed exclusively at ensuring that the Iberian Lynx population is back to the 1990 levels (1,100 specimens). Consider that it is the minimum threshold necessary to ensure its preservation. The money would only be allocated for the described purpose.

Would you be willing to pay an additional 30 EUROS to the income tax already imposed (**this year alone**) in order to finance this project?

☐ Yes (*question16*) ☐ No (*question 17*)

16. (If you answered **YES** to the question15). Would you be willing to contribute **45 EUROS**?

☐ Yes ☐ No

17. (If you answered **NO** to the question 15). Would you be willing to contribute **15 EUROS**?

☐ Yes ☐ No

18. What would be the **MAXIMUM AMOUNT** of income tax increase you would be willing to contribute to the fund?

____________ **EUROS** (*question 23 if zero*)

☐ N/A (*question 23 if the answer to questions 15 and 17 were also NO*)

19. Which of the following affirmations explain better what you you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** (adequate from society's point of view) to guarantee the preservation of the Iberian Lynx.
b) I pay because the preservation of the Iberian Lynx is important for **ME**.

20. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I paid the **REST OF SPANISH CITIZENS** (contributors) would pay the same quantity.
b) I thought that if I paid **REST OF SPANISH CITIZENS** (contributors) would pay, but not necessarily the same quantity.
c) I thought that if I paid even a small quantity, this could be **COMPENSATED** by the **PAYMENT OF OTHER CONTRIBUTORS**.
d) **I DID NOT** take into account the quantity that others would pay.

21. In short, which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** that **ALL SPANISH CITIZENS** (contributors) **PAY** to guarantee the preservation of the Iberian Lynx.
b) I pay because preserving the Iberian Lynx is **SOCIALLY CORRECT** and it justifies this **EFFORT ON MY PART**, even if others pay less.
c) I pay because preserving the Iberian Lynx is important for **ME** and it justifies this **EFFORT ON MY PART**, even if others pay less.
d) None of the above, explain your reasons:

22. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would pay **MORE**; please, indicate the quantity: ………………..
b) I would pay **LESS**; please, indicate the quantity: …………………
c) I would pay **THE SAME**.

(*Go to question 26*)

[*Questions for those who did not accept paying any amount of money in **questions 15-18***]

23. Could you tell us the reason why you did not accept paying any amount of money as increased income tax to finance the preservation of the Iberian Lynx?

24. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would pay.
b) I thought that if I did not pay the **REST OF SPANISH CITIZENS** (contributors) would not pay either.
c) **I DID NOT** take into account the quantity that others would pay.

25. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would continue without paying.
b) I **WOULD PAY**; please, indicate the quantity: …………………

26. On a scale of 1 to 5, you found the content of **question 15** (*show it again*):

1 2 3 4 5
It is very unclear — Very clear

27. On a scale of 1 to 5, do you think it is possible to implement the proposal of establishing a fund introduced in **question 15**?

1 2 3 4 5
Impossible — Very possible

28. On a scale of 1 to 5, to what degree of security did you answer **question 15**?

1 2 3 4 5
I very much doubted — I am very sure

Type 4

15. Suppose that the government would put forward a referendum in order to finance a project aimed exclusively at ensuring that the Iberian Lynx population is back to the 1990 levels (1,100 specimens). Consider that it is the minimum threshold necessary to ensure its preservation. The endorsement of this referendum would imply an income tax increase. The money would only be allocated for the described purpose.

The project would only be launched if a majority voted for it.

Would you vote in favor in this national referendum if the proposed increase implied the payment of an additional **10 EUROS** to the income tax already imposed (**this year alone**)?

☐ Yes (*question16*) ☐ No (*question 17*)

16. (If you answered **YES** to the question15). Would you be willing to contribute **30 EUROS**?

☐ Yes ☐ No

17. (If you answered **NO** to the question 15). Would you be willing to contribute **5 EUROS**?

☐ Yes ☐ No

18. What would be the **MAXIMUM AMOUNT** of income tax increase at which you would vote in favor in the referendum?

____________ **EUROS** (*question 23 if zero*)

☐ N/A (*question 23 if the answer to questions 15 and 17 were also NO*)

19. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** (adequate from society's point of view) to guarantee the preservation of the Iberian Lynx.
b) I pay because the preservation of the Iberian Lynx is important for **ME**.

20. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that if I paid the **REST OF SPANISH CITIZENS** (contributors) would pay the same quantity.
b) I thought that if I paid **REST OF SPANISH CITIZENS** (contributors) would pay, but not necessarily the same quantity.
c) I thought that if I paid even a small quantity, this could be **COMPENSATED** by the **PAYMENT OF OTHER CONTRIBUTORS**.
d) **I DID NOT** take into account the quantity that others would pay.

21. In short, which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I pay because I consider **SOCIALLY CORRECT** that **ALL SPANISH CITIZENS** (contributors) **PAY** to guarantee the preservation of the Iberian Lynx.
b) I pay because preserving the Iberian Lynx is **SOCIALLY CORRECT** and it justifies this **EFFORT ON MY PART**, even if others pay less.
c) I pay because preserving the Iberian Lynx is important for **ME** and it justifies this **EFFORT ON MY PART**, even if others pay less.
d) None of the above, explain your reasons:

22. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would pay **MORE**; please, indicate the quantity: ………………..
b) I would pay **LESS**; please, indicate the quantity: …………………
c) I would pay **THE SAME**.

(*Go to question 26*)

[*Questions for those who did not accept paying any amount of money in **questions 15-18***]

23. Could you tell us the reason why you would not vote in favor of the proposed referendum to finance the preservation of the Iberian Lynx?

24. Which of the following affirmations explain better what you thought while answering **questions 15 to 18**:

a) I thought that although I did not vote in favor the **REST OF SPANISH CITIZENS** would vote in favor.
b) I thought that if I did not vote the **REST OF SPANISH CITIZENS** (contributors) would not vote either.
c) **I DID NOT** take into account what others would vote.

25. If you had the guarantee that **ALL SPANISH CITIZENS** (contributors) would pay the same amount as **YOU** do, then:

a) I would continue without paying.
b) I **WOULD PAY**; please, indicate the quantity: …………………

26. On a scale of 1 to 5, you found the content of **question 15** (*show it again*):

1 2 3 4 5
It is very unclear Very clear

27. On a scale of 1 to 5, do you think it is possible to implement the proposal of establishing a fund introduced in **question 15**?

1 2 3 4 5
Impossible Very possible

28. On a scale of 1 to 5, to what degree of security did you answer **question 15**?

1 2 3 4 5
I very much doubted I am very sure

The following questions are shared by the four types of surveys

In what follows, we will ask you some questions about your visits to forests in Spain.

SHOW BOOKLET (recreation)

29. Have you made **RECREATION VISITS** to natural forests areas at least once in the past 12 months?

☐ Yes ☐ No (*question 43*)

30. Would you know if there were stone pine forests within these areas?

☐ Yes (*question 31*) ☐ No (*question 32*)

31. How many days did you visit these natural stone pine forest areas in the last 12 months?

____

32. Which **NATURAL AREA** did you visit in your **LAST RECREATIONAL VISIT** to a forest (of each type) and **HOW MANY DAYS**?

- Natural area:
- Number of days:

33. How much **MONEY** did the **VISIT** cost you and/or your family in:?

- Gasoline or public transport ________ €
- Tolls: ________ €
- Parking: ________ €

34. Could you tell us if this **MONEY** already calculated **WAS SPENT**?:

☐ Only in **YOUR** visit
☐ In YOUR visit and in the visit of YOUR FAMILY and or FRIENDS (specify the number of people you paid for, including yourself): ______

35. Could you indicate us the **MEAN OF TRANSPORT** you used in this visit?

36. Was the **APPROACH ROUTE** another attraction to this visit?

☐ Yes ☐ No
☐ Only part, specify: ……………………

37. Was the practice of hiking, biking, climbing or any other similar activity the **MAIN REASON** for your visit?

☐ Yes ☐ No

Think now about the recreational visits you make to Spanish forests and consider your gas expenditures and the possibility of the establishment of an entrance fee for forest access instituted by the landowner (this would be valid whether it is a public institution or a private party, since both have the right to exclude access).

We ask you to imagine that in your next recreation visit to a forest you have the opportunity to decide only among the alternatives shown in the following cards (OPTIONS A, B, C). In addition, you would have the option to stay home (OPTION D). Please, indicate in each set which of the alternaitves put forward WOULD YOU CHOOSE (ONLY ONE). Bear in mind that if you chose to pay, you could not spend that money on other things.

Question 38a.	CHOICE SET 1R (code 1)			
	Option A	Option B	Option C	Option D
Type of forest	Stone pine	Cork oak	Stone pine	I prefer to stay at home
Infrastructures for recreation	No	Yes	No	
Domestic animals	Yes	Yes	No	
Picking mushrooms	Yes	Yes	No	
Increase in gas expenses	7 euros	17 euros	12 euros	
Entrance fee for accessing the forest	12 euros	2 euros	17 euros	
MARK ONLY ONE OPTION (A,B,C or D)	OPTION A □	OPTION B □	OPTION C □	OPTION D □

Question 38a.	RANKING SET 1R (code 1)			
	Option A	Option B	Option C	Option D
Type of forest	Stone pine	Cork oak	Stone pine	I prefer to stay at home
Infrastructures for recreation	No	Yes	No	
Domestic animals	Yes	Yes	No	
Picking mushrooms	Yes	Yes	No	
Increase in gas expenses	7 euros	17 euros	12 euros	
Entrance fee for accessing the forest	12 euros	2 euros	17 euros	
RANK THE FOUR OPTION (A,B,C or D)	OPTION A 1ª□ 2ª□ 3ª□ 4ª□	OPTION B 1ª□ 2ª□ 3ª□ 4ª□	OPTION C 1ª□ 2ª□ 3ª□ 4ª□	OPTION D 1ª□ 2ª□ 3ª□ 4ª□

39. Only if you chose OPTION D in both cases: Could you tell us why you would stay at home?

Please indicate your level of agreement/ disagreement with the following statements (1 means 'Completely disagreement ' and 5 'Full agreement')

43. Nature conservation is important because I enjoy visiting natural areas.

1 2 3 4 5
Complete disagreement Full agreement

44. I would like to contribute to causes such as nature protection and, when I can afford it, I contribute my time and/or money:

1 2 3 4 5
Complete disagreement Full agreement

45. I am delighted to hear that a new National Park has been created, as I think that both my children and I can visit it:

1 2 3 4 5
Complete disagreement Full agreement

46. When I see a beggar on the street, I cannot resist giving alms:

1 2 3 4 5
Complete disagreement Full agreement

47. Although I and my family can in no way observe the Iberian Lynx in its natural habitat, I get great satisfaction from knowing that the preservation of this species will be ensured.

1 2 3 4 5
Complete disagreement Full agreement

48. I feel very good about myself when giving money on a donation campaign for a good cause.

1 2 3 4 5
Complete disagreement Full agreement

49. Although I had not heard before of a certain bird species, it pains me to know that this species has become extinct:

1 2 3 4 5
Complete disagreement Full agreement

50. If a volunteer approaches me asking for some money on the street for a recognized organization such as the Red Cross, I donate something:

1 2 3 4 5
Complete disagreement Full agreement

51. Despite the fact that neither my family nor I can see the black vulture in the natural habitat, I am pleased to know that the Spanish legislation protects these birds:

1 2 3 4 5
Complete disagreement Full agreement

52. There are some fundraising/donation campaigns to which I feel very close and I do not hesitate to support with a donation, regardless of which organization promoted the campaign:

1 2 3 4 5
Complete disagreement Full agreement

To conclude, we would like to ask you some personal questions. We remind you that this questionnaire is anonymous.

53. Year of birth

54. Municipality of the habitual residence

55. Marital status (it is not read):

- ☐ Single
- ☐ Married
- ☐ Divorced
- ☐ Widowed

56. Level of education (it is not read):

- ☐ No schooling
- ☐ Primary studies
- ☐ Secondary school
- ☐ Diploma
- ☐ Bachelor
- ☐ Other (specify)

57. Occupation (*it is not read*):

- ☐ Salaried employee
- ☐ Entrepreneur and self-employed worker
- ☐ Unemployed
- ☐ Household tasks
- ☐ Student
- ☐ Retired
- ☐ Other (specify)

58. Do you work in a profession related to the forestry sector?

☐ Yes ☐ No

59. Would you say that the average NET TOTAL INCOME PER MONTH of your family is included in one of these tranches? Remind that this questionnaire is totally anonymous and that we will only use this information for statistical purposes. Show:

I do not have any direct income	
Less than €1,000	
Between €1,001 and €2,000	
Between €2,001 and €3,000	
More than €3,001	
N/A	

60. ¿How many members are in the family unit (including yourself) to whom you referred in the previous question?

61. ¿How many members of this group (including yourself) bring in income to the family unit?

62. Would you have any inconvenience in giving us your telephone just in case we were to make you some additional question? (Record the name, if he/she agrees to give it)

THANK YOU AND HAVE A NICE DAY

TO BE COMPLETED BY THE INTERVIEWER

63. Interviewer:

64. Date:

65. Location of the interview:

66. Start time:

67. Finish time:

68. Respondent's attitude
☐ Reluctant ☐ Indifferent ☐ Good

69. Degree of understanding
☐ Low ☐ Middle ☐ High

70. In your opinion, what part of the survey has been poorly understood by the interviewee?
☐ None
☐ Booklet
☐ Reforestations section
☐ The Iberian Lynx section
☐ Recreational use section
☐ Other (specify):

71. From what question do you consider that the interviewee was fatigued?

72. Did the interviewee have difficulty understanding the type of tax/fund that he/she would have to pay in question15?
☐ Yes ☐ No

73. Gender of the interviewee
☐ Male ☐ Female

74. In your opinion, the survey has become long:
☐ Yes ☐ No

75. Comments:

APPENDIX I: THE SIMULATED EXCHANGE VALUE METHOD WITH TWO AND THREE ALTERNATIVES

The revenue function is:

$$R_j = p_j \cdot q_j\left(p_j\right) - c\left(q_j\right) \quad \text{[I.1]}$$

and considering that

$$c^{'}\left(q_j\right) = 0 \quad \text{[I.2]}$$

and that

$$q_j\left(p_j\right) = Q\,\mathrm{Pr}\left(p_j\right) \quad \text{[I.3]}$$

where Q represents current visits.

SEV-2: PRICE AND REVENUE MAXIMIZATION

The probability of the alternative *j* is:

$$\mathrm{Pr}_j\left(p_j\right) = \frac{q_j\left(p_j\right)}{Q} = \frac{e^{V_j}}{e^{V_j}+1} = \frac{1}{1+e^{-V_j}} \quad \text{[I.4]}$$

where

$$V_j = \alpha_j + \beta_{p_j} \times p_j \quad \text{[I.5]}$$

so,

$$V_j^{'} = \beta_{p_j} \quad \text{[I.6]}$$

The derivative of [I.4] with respect to p_j is:

$$\mathrm{Pr}_j^{'}\left(p_j\right) = \frac{\beta_{p_j} e^{V_j}\left(e^{V_j}+1\right) - e^{V_j}\left(\beta_{p_j} e^{V_j}\right)}{\left(e^{V_j}+1\right)^2} = \frac{\beta_{p_j} e^{V_j}}{\left(e^{V_j}+1\right)^2} \quad \text{[I.7]}$$

The derivative of [I.1] with respect to p_j is:

$$\frac{\partial R\left(p_j\right)}{\partial p_j} = q_j\left(p_j\right) + p_j q_j^{'}\left(p_j\right) - c^{'}\left(q_j\right) = 0 \quad \text{[I.8]}$$

Equivalently:

$$\frac{\partial R(p_j)}{\partial p_j} = Q\Pr(p_j) + p_j Q \Pr'(p_j) = 0$$

$$= \left[\frac{e^{V_j}}{e^{V_j}+1} + p_j \frac{\beta_{p_j} e^{V_j}}{\left(e^{V_j}+1\right)^2} \right] = 0 \qquad \text{[I.9]}$$

$$= e^{V_j} + 1 + p_j \beta_{p_j} = 0$$

Alternatively:

$$V_j = \alpha_j + p_j \beta_{p_j} = \ln\left(-1 - p_j \beta_{p_j}\right) \qquad \text{[I.10]}$$

so,

$$p_j = \frac{\ln\left(-1 - p_j \beta_{p_j}\right) - \alpha_j}{\beta_{p_j}} \qquad \text{[I.11]}$$

Instead of estimating price and revenue maximization by taking the derivative with respect to p_j, as we have done above, it would also be possible to derive the same results by taking the derivative with respect to $q_j(p_j)$. From [I.4], we know that:

$$1 + e^{-V_j} = \frac{Q}{q_j(p_j)}$$

$$e^{-V_j} = \frac{Q}{q_j(p_j)} \quad 1 = \frac{Q - q_j(p_j)}{q_j(p_j)} \qquad \text{[I.12]}$$

$$e^{V_j} = \frac{q_j(p_j)}{Q - q_j(p_j)}$$

so,

$$V_j = \alpha_j + p_j \beta_{p_j} = \ln\left(\frac{q_j(p_j)}{Q - q_j(p_j)} \right)$$

$$p_j = \frac{\ln\left(\frac{q_j(p_j)}{Q - q_j(p_j)} \right) - \alpha_j}{\beta_{p_j}} \qquad \text{[I.13]}$$

Substituting [I.13] in [I.1] and calculating the derivative with respect to $q_j(p_j)$:

$$\frac{\partial R(p_j)}{\partial q_j(p_j)} = \frac{Q}{Q - q_j(p_j)} + \ln\left(\frac{q_j(p_j)}{Q - q_j(p_j)} \right) = \alpha_j \qquad \text{[I.14]}$$

SEV-3: PRICE AND REVENUE MAXIMIZATION

The probability of the alternative *j* is:

$$\Pr_j\left(p_j\right)=\frac{q_j\left(p_j\right)}{Q}=\frac{e^{V_j}}{e^{V_j}+e^{V_y}+1} \qquad [I.15]$$

or equivalently,

$$\frac{Q}{q_j\left(p_j\right)}=\frac{e^{V_j}+e^{V_y}+1}{e^{V_j}}=1+\frac{e^{V_y}}{e^{V_j}}+\frac{1}{e^{V_j}}$$

$$\frac{Q}{q_j\left(p_j\right)}-1=\frac{e^{V_y}+1}{e^{V_j}} \qquad [I.16]$$

$$\frac{Q-q_j\left(p_j\right)}{q_j\left(p_j\right)}=\frac{e^{V_y}+1}{e^{V_j}}$$

where

$$V_j=\alpha_j+p_j\beta_{p_j} \quad \text{and} \quad V_y=\alpha_y+p_y\beta_{p_y} \qquad [I.17]$$

so,

$$V_j^{'}=\beta_{p_j} \qquad \text{and} \quad V_y^{'}=\beta_{p_y} \qquad [I.18]$$

The derivative of [I.15] with respect to p_j is:

$$\Pr{}'_j\left(p_j\right)=\frac{\beta_{p_j}e^{V_j}\left(e^{V_j}+e^{V_y}+1\right)-e^{V_j}\left(\beta_{p_j}e^{V_j}+\beta_{p_j}e^{V_y}\right)}{\left(e^{V_j}+e^{V_y}+1\right)^2}=\frac{\beta_{p_j}e^{V_j}}{\left(e^{V_j}+e^{V_y}+1\right)^2} \qquad [I.19]$$

The derivative of [I.1] with respect to p_j is:

$$\frac{\partial R\left(p_j\right)}{\partial p_j}=Q\left[\Pr\left(p_j\right)+p_j\Pr{}'\left(p_j\right)\right]=0$$

$$=Q\left[\frac{e^{V_j}}{e^{V_j}+e^{V_y}+1}+p_j\frac{\beta_{p_j}e^{V_j}}{\left(e^{V_j}+e^{V_y}+1\right)^2}\right]=0 \qquad [I.20]$$

$$= \quad e^{V_j}+e^{V_y}+1+p_j\beta_{p_j}=0$$

Alternatively:

$$e^{V_j}= \quad e^{V_y}-1-p_j\beta_{p_j}$$

$$V_j=\alpha_j+p_j\beta_{p_j}=\ln\left(-e^{V_y}-1-p_j\beta_{p_j}\right) \qquad [I.21]$$

so,

$$p_j = \frac{\ln\left(-e^{V_y} - 1 - p_j \beta_{p_j}\right)}{\beta_{p_j}} \qquad [I.22]$$

As before, it would also be possible to estimate price and revenue maximization by taking the derivative with respect to $q_j(p_j)$. From [I.16], we know that:

$$e^{V_j} = \frac{q_j(p_j)(e^{V_y} + 1)}{Q - q_j(p_j)}$$

$$V_j = \alpha_j + p_j \beta_{p_j} = \ln\left(\frac{q_j(p_j)(e^{V_y} + 1)}{Q - q_j(p_j)}\right) \qquad [I.23]$$

so,

$$p_j = \frac{\ln\left(\frac{q_j(p_j)(e^{V_y} + 1)}{Q - q_j(p_j)}\right) - \alpha_j}{\beta_{p_j}} \qquad [I.24]$$

Given [I.17], we obtain that [I.24] is now:

$$p_j = \frac{\ln\left(\frac{q_j(p_j) e^{\alpha_y + p_y \beta_{p_y}}}{Q - q_j(p_j)}\right) - \alpha_j}{\beta_{p_j}} \qquad [I.25]$$

Taking napierian logarithm, noting that $e^{\alpha_j} = e^{\ln\left(e^{\alpha_j}\right)}$ and that $\ln\left(e^{\alpha_j}\right) = \alpha_j$, we get that:

$$\ln(q_j(p_j)) + \ln\left(e^{\alpha_y + p_j \beta_p} + 1\right) - \ln(Q - q_j(p_j)) - \ln(\alpha_j) = p_j \beta_{p_j} \qquad [I.26]$$

Considering that:

$$\ln\left(\frac{Qe^{\alpha_j}}{q_j(p_j)} - e^{\alpha_j}\right) = \ln\left(Qe^{\alpha_j} - q_j e^{\alpha_j}\right) - \ln(q_j(p_j)) \qquad [I.27]$$

we derive [I.26] as:

$$\ln\left(\frac{q_j(p_j)\left(e^{\alpha_y` + p_j \beta_p} + 1\right)}{q_j(p_j) e^{\alpha_j} - Qe^{\alpha_j}}\right) = p_j \beta_{p_j} \qquad [I.28]$$

As:

$$\ln(q_j(p_j)) + \ln\left(e^{\alpha_y + p_y \beta_p} + 1\right) - \ln\left(q_j(p_j) e^{\alpha_j} - Qe^{\alpha_j}\right) = p_j \beta_{p_j} \qquad [I.29]$$

[I.29] is now:

$$\ln\left(q_j\left(p_j\right)\right)+\ln\left(e^{\alpha_y+p_j\beta_p}+1\right)-\ln\left(Qe^{\alpha_j}-q_j\left(p_j\right)e^{\alpha_j}\right)=p_j\beta_p \qquad \text{[I.30]}$$

Note that:

$$\ln\left(\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}-e^{\alpha_j}-e^{\alpha_h}\right)=\ln\left(Qe^{\alpha_j}-q_j\left(p_j\right)e^{\alpha_j}-q_j\left(p_j\right)e^{\alpha_h}\right)-\ln\left(q_j\left(p_j\right)\right) \qquad \text{[I.31]}$$

and that

$$\ln\left(\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}-e^{\alpha_j}-\left(\frac{q_j\left(p_j\right)e^{\alpha_y}}{q_j\left(p_j\right)-Q}\right)+\frac{Q}{q_j\left(p_j\right)-Q}\right)=$$

$$=\ln\left(2q_j\left(p_j\right)Qe^{\alpha_j}-Q^2e^{\alpha_j}-q_j^{\,2}\left(p_j\right)e^{\alpha_j}-q_j^{\,2}\left(p_j\right)e^{\alpha_y}+q_j\left(p_j\right)Qe^{\alpha_y}\right) \qquad \text{[I.32]}$$

$$-\ln\left(q_j\left(p_j\right)\right)-\ln\left(Q-q_j\left(p_j\right)\right)$$

Then:

$$p_j=-\frac{1}{\beta_p}\left[\ln\left(\frac{Q}{q_j\left(p_j\right)}-e^{\alpha_y}\right)+\ln\left(-e^{\alpha_j}q_j\left(p_j\right)+e^{\alpha_j}\right)\right]$$

$$=-\frac{1}{\beta_p}\left[2q_j\left(p_j\right)Q\alpha_j-q_j^{\,2}\left(p_j\right)\alpha_j-q_j^{\,2}\left(p_j\right)\alpha_y+q_j\left(p_j\right)\alpha_y-Q\alpha_j-\ln\left(q_j\left(p_j\right)\right)\right] \qquad \text{[I.33]}$$

Considering that $\alpha_j=\ln\left(e^{\ln\left(e^{\alpha_j}\right)}\right)=\ln\left(e^{\alpha_j}\right)$ and rearranging, [I.33] is now:

$$p_j=-\frac{1}{\beta_{p_j}}\left[\ln\left(e^{\alpha_j}q_j\left(p_j\right)-Qe^{\alpha_j}+q_j\left(p_j\right)e^{\alpha_y}\right)+\ln\left(-q_j\left(p_j\right)+Q\right)-\ln\left(q_j\left(p_j\right)\right)\right] \qquad \text{[I.34]}$$

According to [I.31] and [I.32]:

$$p_j=-\frac{1}{\beta_{p_j}}\left[\ln\left(\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}-e^{\alpha_j}-e^{\alpha_y}\right)+\ln\left(-q_j\left(p_j\right)+Q\right)\right] \qquad \text{[I.35]}$$

As both are equal:

$$p_j=-\frac{1}{\beta_{pv}}\ln\left[\left(\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}\right)-e^{\alpha_j}+\frac{q_j\left(p_j\right)e^{\alpha_y}}{Q\quad q_j\left(p_j\right)}-\frac{e^{\alpha_y}}{Q\quad q_j\left(p_j\right)}\right]=\frac{1}{\beta_{pv}}\ln\left[\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}-e^{\alpha_j}-e^{\alpha_y}\right] \qquad \text{[I.36]}$$

Substituting [I.36] in [I.1] and calculating the derivative with respect to $q_j\left(p_j\right)$:

$$\frac{\partial R\left(p_j\right)}{\partial q_j\left(p_j\right)}=\ln\left(\frac{Qe^{\alpha_j}}{q_j\left(p_j\right)}\right)-e^{\alpha_j}-e^{\alpha_y} \qquad \text{[I.37]}$$

APPENDIX J: RESULTS FROM THE SINGLE-BOUNDED AND DOUBLE-BOUNDED MODELS FOR HOMO STRATEGICUS

We extend the dichotomy between individuals who act as free-riders and those who do not follow a strategic behavior, regardless of whether they are willing to pay or not. Thus, total sample is now made up of 657 individuals (596 individual who pay and 61 who do not definitely pay – excluding protest responses), as shown in Table 5.2. Once classified, the variable FR is dummy-coded (1 if respondent is not FR and 0 if respondent is FR).

Mean parameters from single-bounded and double-bounded models are not significant (Table J.2) for *FR*. While theory predicts the free-rider problem, it seems that when respondents face a real scenario, they prefer not to reveal their behavior. The variable *ASC-REC* shows a positive sign, meaning a preference for contributing in order to ensure the preservation of the Iberian Lynx. The variable *Bid* also offers negative sign. The variable *FR* has positive sign, so respondents are more likely to show a preference for not being a free-rider rather than for having a strategic behavior.

Table J.1. Results from the single-bounded and double-bounded models for *Homo Strategicus*

Attribute	Single-bounded	Double-bounded
	Mean parameter	Mean parameter
Intercept	2.0291***	2.2802***
	(0.3094)	(0.2365)
Homo Strategicus (FR)	0.3760	0.2018
	(0.2602)	(0.2232)
Bid	-0.0685***	-0.0797***
	(0.0076)	(0.0366)
Log-likelihood	-382.10	-850.45
Adj. McFadden ρ^2	0.1081	
n	657	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.

When including *Clarity*, *Credibility* and *Certainty*, results indicate that only *Credibility* is significant in the single-bounded model (Table J.2). *FR* offers a negative sign.

Table J.2. Results from the single-bounded and double-bounded models for *Homo Strategicus* using the questions related to the clarity, credibility and certainty of the contingent valuation scenario

Attribute	Single-bounded	Double-bounded
	Mean parameter	Mean parameter
Intercept	1.9137***	2.0266***
	(0.6144)	(0.4337)
Homo Strategicus (FR)	-0.4041	-0.2342
	(0.2630)	(0.2234)
Clarity	0.3731	0.3394
	(0.3731)	(0.2936)
Credibility	0.6344***	0.4499***
	(0.1806)	(0.1474)
Certainty	-0.1682	0.0084
	(0.5486)	(0.3871)
Bid	-0.0674***	-0.0819***
	(0.0078)	(0.0038)
Log-likelihood	-367.51	- 830.03
Adj. McFadden ρ^2	0.1208	
n	657	

Note: standard errors are shown in brackets; n: number of observations; asterisks (e.g.,***,**,*) denote significance at the 1%, 5% and 10% level, respectively.