Provision cost function of forest biodiversity protection within French Natura 2000 network

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Abstract

Biodiversity protection is the outstanding ecosystem service both from ecologic, economic and social point of view. European Union has launched since more than two decades an ecological network of sites known as Natura 2000 the biggest worldwide to enhance biodiversity protection in different habitat types. The Natura 2000 network maintains both economic and conservation activities and provide a realist framework for estimating provision cost of biodiversity protection. The main goal of this research was to estimate provision cost function of biodiversity protection within the Natura 2000 network as a function of the net impact of conducted actions on conservation score. The French Natura 2000 network was selected because of its accessible financial and ecological database as well as access and language facility. A standardized and consistent evaluation method for all French forest habitats of Community interest (Carnino method) was used to assess degree or score of biodiversity conservation before and after implementing biodiversity actions at intervention level. According to this methodology, the most important actions affecting degree of biodiversity conservation have been selected. Selected actions were the following: eliminating or limitation of invasive species (F22711), enhancing senescent (large) trees (F22712) and Marking, felling or pruning without production objective (F22705). These actions are implemented in response to compensation payment within contract systems. More than half of forest contracts using these actions were highly concentrated in five eastern regions (Rhone-Alpes, Alsace, Bourgogne, Champagne-Ardennes and Franche-Comté) of France. In order to respect a minimum time span of 6 years after implementing biodiversity actions, all relevant contracts implemented in the period 2002–2007 have been considered. As such, 36 intervention parcels, 28 contracts within 15 Natura 2000 sites in 5 regions have been selected. The French financial and ecological database of Natura 2000 databases include financial and biological data on pre-treatment or pre-intervention state of sites while biological data on post-treatment or post-intervention states is lacking. As such, biological data on post-treatment status of intervention parcels were collected using telephone survey (questionnaire) and cost-effective plot-less inventory. The annual amount of the contract per hectare of intervention parcel has been calculated without discounting since the compensation payment is made regularly and without discounting. Then, an inverse linear dose-response function and Difference in Difference (DID) method has used to relate provision cost (treatment dose) to the change of conservation score (treatment impact), the type of biodiversity actions (treatment type) and initial region or site specific effects as well as initial conservation score (pre-treatment variables) at intervention level. Given the problem of endogeneity, the model has been then estimated using nonlinear GMM (General Method of Moment) estimator. Results revealed that restoration actions conducted within French Natura 2000 network were significantly effective in promoting degree of conservation of biodiversity at 1% level. The efficiency of these actions in promoting degree of conservation was estimated to be 1%. As such, 1% raise in biodiversity costs will result in 1% increase in conservation status. However, for the same change percent of the degree of conservation, action F22705 costs almost half compared to action F22711. Finally, the average provision cost to enhance by 1% the conservation score of biodiversity through both actions was estimated at 15.6 € ha^{-1} y^{-1}. Similarly, maintenance action (F22712) conducted within French Natura 2000 network were significantly effective in maintaining the degree of conservation of biodiversity at 1% level. The average provision cost to maintain the conservation score of biodiversity through action F22712 was estimated to be 1.4 € per tree per year. However, these results must be interpreted as a short term assessment of biodiversity actions indicating first impacts of biodiversity actions. Long term monitoring of biodiversity actions and their impacts with larger data set will enable a more complete and precise evaluation of intervention impacts. Generalizing this pilot research to include other biodiversity actions and other EU countries will provide relevant information for policy and decision makers.

Key words: Biodiversity, Conservation status, Impact assessment, Natura 2000 site, Provision cost, General Method of Moment

1. Introduction

Forests are the source of a wide range of goods and services of crucial ecological, social and economic importance to society. Broadly defined and classified, forest goods and services are simply the benefits people derive, directly or indirectly, from forest ecosystems functions (MEA, 2005; Mavsar et al., 2008; Stenger et al., 2009). For instance, covering 44% of the total land surface (FAO, 2006) and more than half of the 26400 Natura 2000 sites in Europe (Romao et al., 2012), forest habitats conserve almost 90 percent of the world's terrestrial biodiversity (The World Bank, 2004). Biodiversity is, in turn, a support system for all forest goods and services (MEA, 2005; WWF, 2010). Most of these services do not have a market price to reflect their values or benefits for society. This may be why valuable forests are replaced with other land uses or abandoned or even undermanaged, that is, they are not managed to provide socially desirable quantity/quality of non-market goods and services (Mavsar et al., 2008). Finally, due to this under-valuation, the forests sector attracts less investment than it deserves (Kengen, 1997). Forest valuation has been developing to bridge the gap caused by the absence of market. Valuation looks at the monetary value of a given change in the quantity and/or quality of a forest service along the demand curve (Mavsar et al., 2008). Methodological intricacies and debates on forest valuation have been widely dealt with in the literature. Thus, the methodology and knowledge on non-market valuation methods have improved considerably. These methods value all forest goods and services in a segregated way. Actually, the best practice standards of valuation as those developed under COST action E45 (Navrud, 2010; Thiene et al., 2011) guide usefully their application. Thus, there seems to be numerous databases of soundly estimated economic values covering all forest goods and services. These are even well organized through worldwide, web-based systems such as EVRI (Stenger et al., 2009). Moreover, innovative methods of forest valuation handling the jointly produced externalities in an integrated way are currently under development (NEWFOREX, 2009).

Research problem

While information (concept, methodology etc.) about the values of forest goods and services (demand side) seems to be widely available, the costs of provision (supply side) are often unknown. Indeed, despite the well-known standard management practices for wood production, there are often no recommended managerial practices (actions and measures) to enhance Non Wood Forest Products (NWFPs). Thus, there is little information about provision cost of forest services as well. In addition, estimating provision cost requires assessing the impact of costs on biodiversity conservation. However, there is a gap in the scientific literature about costs of provision. However, in order to enhance the provision of non-market forest services, policy makers (to determine compensation payment and to design an efficient payment mechanisms) and forest managers (to decide and then to apply certain actions or adapt their management practices) need to know both about their values and provision costs. In addition, a good estimation of provision cost is a crucial piece of information for developing markets for these goods and services (NEWFOREX, 2009). Finally, this information will be quite helpful for evaluating and re-designing European programs and funds such as Natura 2000 funds under Rural Development Programs throughout the EU.

Research questions and objectives

In view of the state of the art in forest valuation literature and actual scientific needs in this domain, a great research question can be identified as follows: how to estimate the actual provision cost of a forest service taking into account the net effect of conducted actions on the quantity and/or quality of a forest service. Is it possible to identify the most efficient actions or measures for improving a forest service?

In the pool of forest goods and services, biodiversity conservation is an outstanding service both from ecologic (MEA, 2005), economic (Kengen, 1997) and social (Mavsar et al., 2008) point of view in Europe as in the World. Although comparative estimates of the value of forest goods and services at an EU level are difficult, EU level experts quote biodiversity protection (biospheric services) as being the most important forest service in Europe (Mavsar et al., 2008). Moreover, biodiversity seems to be among the first forest services targeted not only by valuation researches (Stenger et al., 2009) but also more recently by compensation payments within Natura 2000 network under Habitat Directive and Rural Development Program at EU level (Rekola, 2003; Marage and Delmas, 2008). This is why this research focuses on biodiversity protection rather than other forest goods and services.

As such, the main goal of this research was to estimate the provision cost function of biodiversity protection within the Natura 2000 network based on the net impact of conducted actions on biodiversity conservation.

Monitoring and impact assessment of the Natura 2000 program and actions on the conservation of biodiversity are required both under article 17 of the Habitats Directive (2007–2012) and article 80 of Council Regulation (EC) 1698/2005 for RDP (2007–2012). However, these assessments are both implemented at large scales (at program or bio-geographic region and national level) and do not explicitly deal with the provision cost of biodiversity protection. However, micro level assessment of impacts of conducted actions on biodiversity conservation seems to be more feasible especially taking into account the multiplicity of variables affecting biodiversity conservation (e.g., pre-treatment, context and treatment variables), which expand as the level goes up. Moreover, the micro-level analysis seems more reliable as the detailed and non-aggregated information on pre-treatment conditions, conducted actions, elapsed time and its new condition are available. As a result, provision cost of biodiversity conservation is a function of the net impact of conducted actions on the conservation of biodiversity at intervention level. In so doing, the effectiveness and efficiency of biodiversity actions will be also presented and discussed. As such, the identified problematic and objective seems to be fully relevant and original both from practical and scientific point of view. In addition, it seems to be in line with the EFI's main mission of providing unbiased and policy relevant information on European forests and forestry (See the host institution and the context of study).

The host institution and the context of the study

One of the main aims of the European Forest Institute (EFI) is to provide researchers, decision makers and the public unbiased, research-related and policy-relevant information on European forests and forestry. The Observatory for European Forests (OEF) supports EFI in this work. OEF research projects are in line with the EFI Research and Development strategy and support EFI in pan-European/global research issues particularly in the field of policy and economics. In this context, OEF supports different projects.

This project has been defined and conducted by Dr. S.M. Heshmatol Vaezin, senior researcher at OEF and assistant professor of the University of Tehran, during the period from July 2011 to August 2012. The research draws on the collaboration, expertise and contributions of a large number of people within EFI, INRA, AgroParisTech (Forest Economics and forest-wood resources laboratories) and the French Natura2000 network. The overall objective of the project is to improve the information about provision costs of major forest services in Europe. Comparable cost and value information (i.e., cost and value of a marginal change in the quantity and/or quality of a service) for different forest services in Europe will be quite helpful for decision and policy makers in different levels (e.g., for compensating or optimizing the provision level of services) especially for evaluating and re-designing European programs and funds (e.g., Water Framework Directive, Habitat Directive, Rural Development Regulation). In this context, a pilot study (provision cost of a forest service in an EU country) is conceived as a methodological model to be used for different forest services in EU countries.

2. Sources of evidences

The question related to sources of evidences is also very relevant not only from a methodological point of view but also for defining the scope of the research i.e., describing precisely the object, suitable program and/or sites for study.

2.1 Studied forest service

Forest goods and services can be classified according to different classification factors. the forest functional classification is an ecosystem-based classification starting from the ecosystem functions while the Total Economic Value (TEV) classification is a use-based approach classifying the goods and services according to how society benefits from them (Mavsar et al., 2008). According to the functional classification of MEA (2005), biodiversity protection is a biospheric service of forests. By contrast, according to TEV classification

(Kengen, 1997; Mavsar et al. 2008), society benefits from biodiversity protection as a forest service through its indirect uses, eventual future uses (option value) and non-use benefits (existence value and altruism). Biodiversity protection involves protecting of biophysical (e.g., habitat) and biological entities (e.g., species). Although biodiversity protection is considered as a forest service, but biodiversity, as biologic entities, is not a service but rather the support for all ecosystem goods and services (MEA, 2005). While the information about the value of biodiversity seems to be sufficiently available, there is, however, a great gap in the literature about its provision cost. While forests are often not managed to provide socially desirable quantity/quality of non-market goods and services (Mavsar et al. 2008), sound information on the provision costs seems to help their provision (e.g., via incentive contracts). Biodiversity protection is the outstanding forest service both from ecologic (MEA, 2005), economic (Kengen, 1997) and social (Mavsar et al., 2008) point of view in Europe as in the World.

2.2 Selecting suitable scheme/program

The improved practices and measures enhancing biodiversity or other forest services have most often been implemented in response to financial incentives or payment mechanisms under different schemes (e.g., agri-environment) or programs (e.g., RDP). While many market mechanisms are often still in the project/starting phase, public payment schemes prevail by far (SFC, 2008). Although many States also use State aids, the main public funding mechanism in European Union for supporting the provision of non-market services are being provided under the European Agricultural Fund for Rural Development (EAFRD) which finance EU's Rural Development Program, RDP (2007–2013) for each European State members. RDP offer a broad range of measures potentially capable of providing compensation payments for the provision of forest services. Member States can choose from eight RDP measures specifically for forestry. In this package, at least three measures aim at improving non-market services via payment for certain improvements in the management practices of existing forests (e.g., support for non-productive investments, Natura 2000 compensation payments and Forest-environment payments). These compensation payments are estimated based on the provision costs of different non-market services.

The European Network for Rural Development (ENRD) diffuses information about implementation and evaluation of EU's Rural Development Programs. This network summarizes annually all compensation payments by axis and measure for member States in the European Union (AGRI H4, 2009). Consequently, providing detailed information in terms of financial and common impact indicators both at measure and program levels, ENRD can be seen as a very good European data platform for estimating provision cost of forest services. However, public financial payments for forest services under Rural Development Regulation (1698/2005 EC) is comparably new and hence have had low uptake in the member States of the European Union (SFC, 2008). In addition, it is widely recognized that in most cases, it is too early for a true assessment of impact of recent forestry measures especially on the quantity and/or quality of forest services (in most case from 2007 onward). This is why, in most of the cases, little quantitative information can be found on the mid-term evaluation (2010) of the forestry measures. Consequently, estimation of provision cost of major forest services using this information source does not seem actually to be possible but is of an excellent potential for the future researches.

By contrast, Biodiversity protection is the centrepiece of EU nature and biodiversity policy and part of the requirements of the UN Convention on Biological Diversity. Biodiversity protection seems to be the objective of several successive programs and funding at European level from the Habitat Directive to EAFRD/Rural development program (Axe 2, measure 224 called, Natura 2000 payment). An EU-wide ecological network of sites known as Natura 2000 was established under the Habitats Directive (1992/43/EEC) to protect land areas for biodiversity. As outlined in Article 2 of Directive "Habitats, Fauna, Flora," the objective of the development of Natura 2000 sites is not to put the nature "under glass". In contrast, Natura 2000 network aims to "maintain or restore species of community importance and their natural habitats in a favorable conservation status "while" taking account of economic, social and cultural requirements and constraints as well as regional and local characteristics. In this context, the Habitats Directives finance eligible actions to enhance biodiversity protection. The practical approach of Habitats Directives in Natura 2000 program i.e., maintaining both economic and conservation activities in progress (Instead of banning all non-conservation activities), provide a realistic framework for estimating provision cost of biodiversity protection. Moreover, the ambitious ecological network of "Natura 2000" sites is the first of its kind that has launched since almost two decades by the European Union, (Demoly, 2010). Natura 2000 sites protect more than a thousand species and 231 habitat types and cover 18% of EU land area (EUSTAFOR and Patterson, 2011). Consequently, data on biodiversity protection under Natura 2000 network seems actually the main common, reliable and available information about biodiversity protection in Europe. Natura 2000 sites may cover forests, agricultural lands, or other land cover. Accordingly, there are three types of Natura 2000 contracts; forest contracts, agricultural contracts, non-forest, non-agricultural contracts. In this research, the focus has been given to forest Natura 2000 sites (and contracts) containing at least one type of forest habitat.

2.3 Study sites

The pilot study of French Natura 2000 sites was justified because of the French model (Contract system), experiences and comprehensive database (within Natura 2000 network). In addition, the study of the French network as a second largest Natura 2000 network in Europe was justified due to access and language facility.

During the period 2002–2007, 99 Natura 2000 contracts have been signed and implemented in France. Forest contracts employ different biodiversity actions. However, three biodiversity actions including eliminating invasive species (F22711), maintaining very large living trees (F22712) and increasing deadwood (F22705) appeared to be the most often used in forest contracts. 57 forest contracts out of 99 contracts i.e., almost 58 percent of all contracts during the period 2002–2007 targets selected actions (F22705, F22711, F22712). In addition, these are the most important actions affecting the status of biodiversity conservation. As such, these 3 actions have been selected in this study. All forest contracts employing the selected actions were highly concentrated in certain regions. Indeed, 54 percent of these 57 contacts (i.e. 31 contracts) are located only in the two French regions of Rhone-Alpes (35%) and Alsace (19%) while the remaining 46 percent of contracts are located in 16 French regions (i.e., <3% of contracts per region). This concentration facilitates the second evaluation of conservation status. Consequently, both regions of Rhône-Alpes and Alsace have been selected as a study site. However, after accomplishing a telephone survey, only 22 contracts and 28 intervention parcels revealed to be relevant or feasible for this research. To respect a minimum set of 30 observations for statistical analysis, all neighbour regions between Alsace and Rhône-Alpes regions have been also selected. As such, three more regions including Bourgogne, Champagne-Ardennes and Franche-Comté with 6 supplementary forest contracts and 8 intervention parcels have been added to the study sites. As such, the study sites cover 36 intervention parcels under 28 forest contracts within 15 Natura 2000 sites in 5 eastern French regions (Table 1).

Table 1 summarizes characteristics of studied Natura 2000 sites. Similarly, Figure 1 shows the study sites on the map of France. The map of Natura 2000 sites has been constructed using the exact coordinates (latitude and longitude) of the centre of Natura 2000 sites (on FSD) through Google Map.

Table 1. Characteristics of studied Natura 2000 sites in the regions Rhone-Alpes, Alsace, Bourgogne, Champagne-Ardennes and Franche-Comté, France.

Site code	Site name	Number of visited contracts	Number of intervention parcels	French Region
FR4202002	VOSGES DU SUD	**11**	**11**	Alsace
FR2600988	HETRAIE MONTAGNARDE ET TOURBIERES DU HAUT MORVAN	**1**	**1**	Bourgogne
FR2100301	FORET DU MONT-DIEU	**1**	**1**	Champagne-Ardennes
FR2100273	TOURBIERES DU PLATEAU ARDENNAIS	**1**	**1**	
FR4310027	LAC DE REMORAY	**1**	**2**	Franche-Comté
FR4301342	VALLEE DE LA SAONE	**1**	**2**	
FR4301348	FORETS ET RUISSEAUX DU PIEMONT VOSGIEN DANS LE TERRITOIRE DE BELFORT	**1**	**1**	
FR8201764	BOIS DE LESPINASSE, DE LA BENISSON-DIEU ET DE LA PACAUDIERE	**1**	**1**	Rhône-Alpes
FR8201670	CEVENNES ARDECHOISES	**2**	**3**	
FR8201741	FORETS DE RAVINS, LANDES ET HABITATS ROCHEUX DES UBACS DU CHARMANT SOM ET DES GORGES DU GUIERS MORT	**1**	**3**	
FR8201749	MILIEUX ALLUVIAUX ET AQUATIQUES DE L'ILE DE LA PLATIERE	**2**	**3**	
FR8202002	PARTIE ORIENTALE DU MASSIF DES BAUGES : SIC	**1**	**3**	
FR8201686	PELOUSES, FORETS ET GROTTES DU MASSIF DE SAOU	**1**	**1**	
FR8201688	PELOUSES, FORETS ET HABITATS ROCHEUX DE LA MONTAGNE DE L'AUP ET DE LA SARCENA	**1**	**1**	
FR8212006	PERRON DES ENCOMBRES	**2**	**2**	
15 sites		**28 Contracts**	**36 parcels**	**5 regions**

Figure 1. The map of study sites in the five selected regions, France.

3. Literature review on impact assessment

3.1 Impact and impact indicator

Any intervention (program or measure) provides certain impacts. They can be defined as effects of the intervention lasting in medium or long term (DG AGRIa, 2006). By contrast, intervention outputs (accomplished actions) which are immediate exchanges for the support granted (funding) can be seen as very short-run or immediate effects. Similarly, intervention result as immediate advantages (or disadvantages) of direct beneficiaries when an action was completed, can be also considered as a short-run effects. Program impacts have often environmental and socio-economic dimensions. They may be, direct or indirect, positive or negative, expected or unexpected. Moreover, impact scale can be at intervention level or at broader scale. Impact indicators are simply a summary expression of effects in a single number

(Richards and Stokes, 2004). They are quantitative yardsticks measuring environmental and socio-economic benefits of an intervention beyond the immediate effects on its direct beneficiaries (results) both at the level of the intervention but also more generally in the program area (DG AGRI, 2006a). Impact indicators represent measureable yardsticks of the success or failure of a program. Impact indicators are most often linked to the objectives of an intervention (program or measure). Indeed, appropriate impact indicator should be capable of capturing intended (objective related) effects of a given program. For instance, as the main objective of Natura 2000 program is to halt biodiversity decline, maintain biodiversity and enhance biodiversity protection (environmental objective), environmental impacts related to biodiversity must be privileged. In order to be used in cost effectiveness and efficiency analysis, impact indicators should be expressed in an absolute value. Moreover, the use of quantitative indicators has some benefits like their measurability, the ability to aggregate data, to repeat measurements after a certain time span and to compare them to earlier ones. Finally, impact indicators should be first in relation to those (people, farms, sites, firms) directly affected by intervention (micro level) before upscaling to higher level (Lukesch and Schuh, 2010). In this context, objective related baseline indicators at the level of Natura 2000 sites seems interesting as they serve both as a baseline (or reference) and impact indicators. In fact, the net change in a given objective related baseline indicator over time due to an intervention such as a Natura 2000 contract can be considered as an impact indicator (DG AGRI, 2006a).

Baseline indicators define the state of the economic, social or environmental situation (both at the level of the intervention but also more generally in the Program area) over time, especially at the beginning of an intervention. Baseline indicators may be related to the objective of a program (objective-related) or to the socio-economic and environment context (context-related) in which the program will be implemented.

3.2 Impact assessment methods

Impact assessment is the process of gauging ‘’net’’ impact or additionality of an intervention. When an intervention with a given objective targets any well-defined “economic, social or territorial unit” (e.g., enterprises, individuals like individual framer or site owner, urban or rural communities or territories), the gross impact of the intervention can be gauged by the difference between relevant indicators (objective related baseline indicator) of factual (treatment groups) and counterfactual (control or comparison groups) situations. Factual situation is the real situation of a group of units (individual farms or sites, firms or enterprises, community or territory, etc.) affected by or exposed to an intervention or a treatment, called treatment group. By contrast, counterfactual situation refers to a hypothetical state of the same treatment group, which would have arisen if a given intervention or program ("policy-off") had not taken place (DG AGRI, 2006b; Evalsed, 2012). In other words, physical impacts of a program such as Natura 2000 must be evaluated relative to what would have arisen in the absence of such a program, referred to as the baseline scenario or counterfactual situation (Richards and Stokes, 2004). For instance, what would have happened to the respective Program area without a given Program? As such, we can never observe a hypothetical counterfactual (without the Program) situation, i.e., the situation of the same treatment group without intervention since the intervention or program had effectively implemented. Thus, the impact of an intervention cannot be seen with certainty (Lukesch and Schuh, 2010). In the practice, counterfactual situation can be considered as the situation of a similar group to treatment group that were not exposed to an intervention, called comparison or control groups. After evaluating factual and counterfactual situations, the mean difference of the relevant indicators on both comparison and treatment groups (e.g., Natura 2000 sites) is the gross impact of intervention. Finally, the gross impact should be net out taking into account the effects of other intervening factors. As such, assessment of impacts consists in two general steps, evaluating factual and counterfactual situations and estimating net impact. Figure 2 synthesizes the procedure and major quantitative methods used for impact assessment.

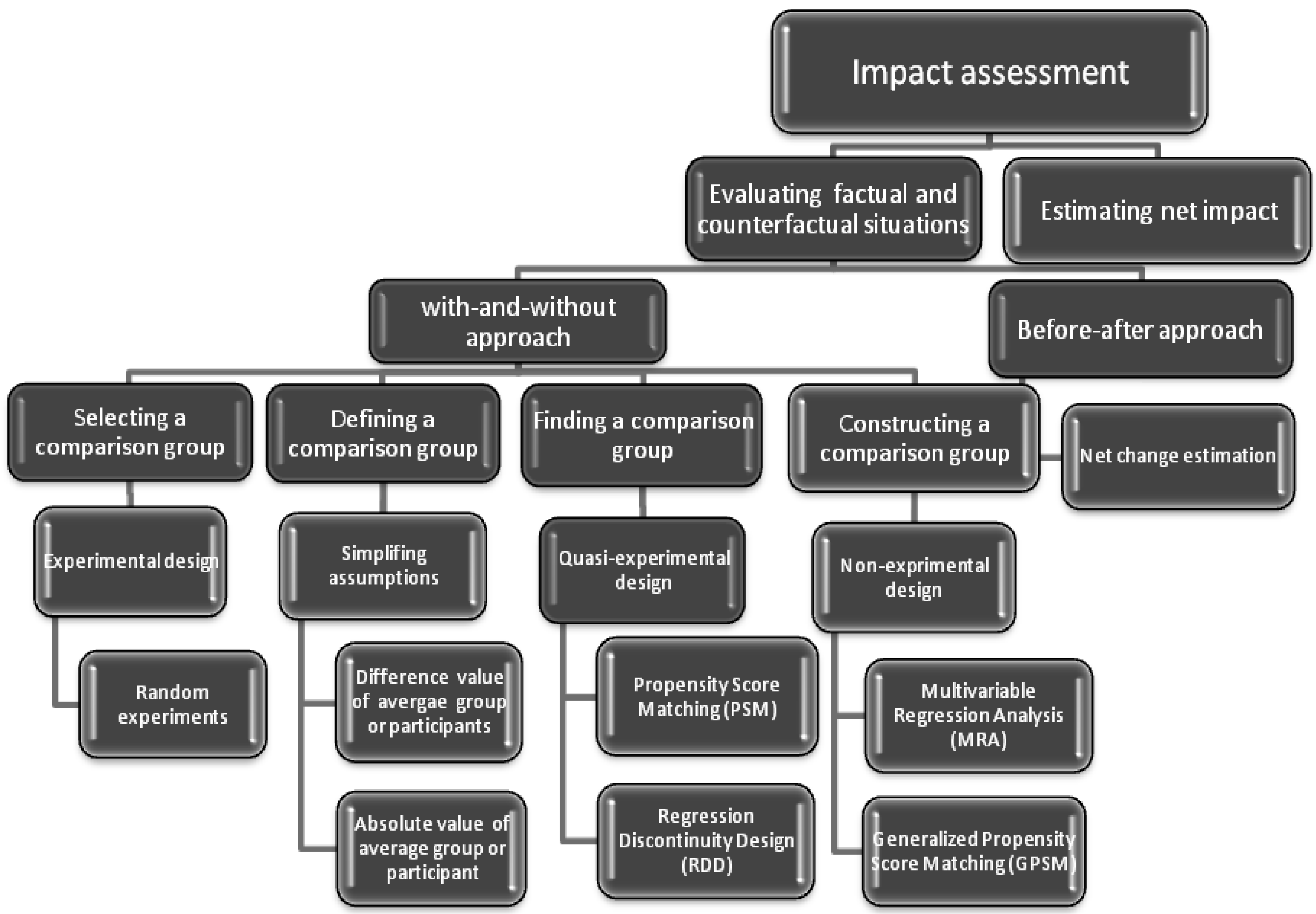


Figure 2. Impact assessment procedure and quantitative methods

The procedure and major methods used for impact assessment will be briefly dealt with in the following sections. Factual and counterfactual situations can be evaluated through two major approaches; ''with and without'' or control group approach and ''before-after'' approach. The both approaches deal similarly with factual situation through evaluating actual status of treatment group. Evaluation of factual situations deals with collecting relevant data at relevant time, especially that of baseline and impact indicators on sites or units affected by intervention (treatment group). This is likely to use existing data or to involve new surveys. By contrast, ’’with and without’’ or control group approach and ‘’before-after’’ approach evaluate counterfactual situation using totally different methods.

3.2.1 ‘’With and without’’ approach

‘’with and without’’ approach evaluates the net impact comparing the situation of treatment group (with intervention or factual situation) and comparison group (without intervention or counterfactual situation). By definition, the only difference between the situations of comparison and treatment groups at a given time comes from the intervention. Thus, the net impact can be estimated without having to net out the observed change of the effects of other direct and indirect intervening factors as well as general trends. Thus, as soon as it is possible, the first best solution is to evaluate without situation using suitable methods:

- Selecting a comparison group (experimental design) if it is already established;
- Defining a comparison (control) group based on the available data (simplifying assumptions);
- Finding a comparison group (quasi-experimental design) and
- Constructing comparison group (Non-experimental design).

The following sections deal with these methods.

Selecting a comparison group (experimental design)

In simplest cases, such as in research laboratories or farms, experimental comparison groups are often well designed and established through methods of experimental design before any intervention (experimental design). In such case, a randomly selected comparison group will be sufficient to evaluate counterfactual situation. However, most of the time we face observational studies in which a group of units is exposed to a well-defined treatment, but unlike an experiment no experimental design are used to maintain a control group. For instance, this predefined comparison groups do not often exist in the realm of public interventions such as Natura 2000 program. Thus, this report does not go further into this matter.

Defining a comparison (control) group

Most of the times, no already established comparison groups are available and thus it is not possible to select randomly a comparison group. By contrast, finding or constructing a comparison group is a time-consuming task especially when the relevant data is lacking. However, a quick and primary evaluation of impact seems to be often helpful. In this context, certain authors define comparison groups based on certain simplifying assumptions (Lukesch and Schuh, 2010). For instance, average situation of all non-participating sites can be defined as a comparison group. However, this definition of comparison group is based on a wrong assumption, considering that average pre-intervention situations (affecting impact indicator) of participating and non-participating sites are the same. As such, this definition of comparison group can lead to substantial errors. Similarly, defining the average situation of all participating and non-participating sites as a control group may not be correct since the average situation of all sites is not necessarily identical to pre-intervention situation of participating sites (Lukesch and Schuh, 2010). To overcome this pre-intervention differences between treatment and comparison groups, certain authors use the before-after change of situation (e.g., in terms of baseline indicators) in non-participating sites as a control groups and compare it to corresponding change in the situation of participating sites (Lukesch and Schuh, 2010). If pre-intervention differences (selection bias) between participating (treatment group) and non-participating sites remains time invariant (‘’fixed” differences over time), comparing before-after changes in relevant indicators of the both groups eliminates this pre-intervention difference and common general trends. Indeed, DID remove systematic pre-intervention differences (fixed effects) between program participants and non-participants which are constant over time. Consequently, before-after changes in the both participating and non-participating sites in the absence of a program or an intervention appear to be identical, as they have fixed pre-intervention difference and feature similar development path (or common trends). As such, this technique named difference in difference (double difference), DID, estimate the net impact of the program. However, the hypothesis of fixed difference or selection bias is not often the case. The selection bias can be statistically tested applying balancing property tests (i.e., testing the similarity of covariates) on panel data covering relevant indicators of participating and non-participating sites. However, in the case of biodiversity protection, collecting relevant indicators such as conservation status in both participating and non-participating sites over an enough longue time and testing selection bias seems to be actually impossible. Nevertheless, the principle of difference in difference (comparison of changes) mitigating pre-intervention differences between participating and non-participating groups remain quite interesting. Moreover, DID permit to net out the estimated impact from any common general trends in both groups. This is why this method can be used in the framework of an experimental, quasi-experimental and non-experimental design (Lukesch and Schuh, 2010). In order to reduce the problem related to the non-validity of the fixed difference hypothesis, the selected non-participating (comparison group) and participating sites must be comparable in essential characteristics (quasi experimental design) or their differences are taken into account through advanced statistical modelling (non-experimental design). In other words, the solution is either to find a comparison group through matching methods or to construct it using statistical modeling.

Finding a comparison group (quasi-experimental design)

The underlying principle of the quasi-experimental design is to find a comparison group (comparable in terms of observable characteristics) from a sample of non-participating sites closest (comparable) to the participating sites through matching methods. When a sample of non-participating sites is available, matching can be done using two major methodologies; Propensity Score Matching (PSM) and Regression Discontinuity Design (RDD). First methodology called PSM use propensity score, PS, as the criteria for finding matching groups. PS is defined as the predicted conditional probability of participation in a program (or assignment to the treatment group) given a set of observed pre-treatment characteristics. PS is estimated as a function of individual characteristics based on a statistical model (e.g., logistic or probit regression). Using the propensity score amounts for matching or subclassifying is much easier than matching or subclassifying across large number of covariates (Imai and Dyk, 2004). After identifying a suitable comparison group, the mean difference of the relevant indicators on both matched comparison group and participating sites or treatment group (e.g., Natura 2000) is the impact of intervention. However, the comparison and treatment sites are not necessarily identical in terms of relevant indicators (e.g., conservation status for Natura 2000 sites) but rather in terms of participation probability. Indeed, applying PSM, it is assumed that all sites having the same predicted probability of participation (close to 1) are similar and thus, selection bias (due to pre-intervention differences between participants and non-participants) should be minimized. Thus, before and after intervention data for both comparison and treatment sites will also be required to account for eventual pre-intervention difference. As such, the combination of PSM and DID methods seems to be useful. The number of observation for treatment sites (beneficiaries) should be large enough to meet statistical requirements, usually 50 observations. By contrast, for non-participating sites (non-beneficiaries), the number of observation should be at least 4 to 10 times higher than that of treatment sites (Lukesch and Schuh, 2010). Moreover, the higher differences between the both sites are, the larger should be the sample size of non-participating sites. The propensity score methods are generally confined to binary treatment regimes or scenarios, i.e., exposed or non-exposed to a treatment. Thus, PS or assignment probability relies on this binary treatment. However, treatment regimes need not be necessarily binary (Imai and Dyk, 2004).

Similarly, the second method, Regression Discontinuity Design (RDD) uses the same principle, i.e., finding a matched comparison group, but using an eligibility threshold on the basis of one or more criteria. As such, all sites (or peoples, farms, firms) classed above this threshold are considered as treatment groups against comparison group being below it. RDD may be a pertinent method only where there are clear and limited rules for sites selection. However, even in this case, selected sites especially in Natura 2000 program cannot be assumed as identical. Moreover, the RDD method assesses only the marginal impact of the Program close to the eligibility threshold (Lukesch and Schuh, 2010). However, for sites with particular or unique characteristics, it becomes difficult to find close matches as a comparison group. In these cases, matching methods do not seem to work properly.

In general, selecting a subset of comparison group (composed of comparison units) similar to the treatment group is difficult because they must be compared across a high-dimensional set of pre-intervention characteristics. In addition, selecting control group involve the risk of selection bias (Kleijn and Sutherland, 2003). For instance, the control group may have a better or worse situation compared to treatment group.

Constructing a comparison group (Non-experimental design)

Pre-defined control groups are often rare in the practice. Moreover, due to the full coverage of certain programs, there is occasionally few or even no subset of program non-participants. Even when there are enough program non-participants, selecting a subset of comparison group (composed of comparison units) similar to the treatment group is difficult because they must be compared across a high-dimensional set of pre-intervention characteristics (Dehejia and Wahba, 2002). Finally, program participants are sometimes unique (e.g., Natura 2000 sites). Thus, comparison groups cannot be randomly selected (experimental design), nor to be identified among program non-participants using matching methods (quasi-experimental design). In this context, the only possible means is to construct a non-experimental comparison group through two major methods; generalized propensity score matching (GPS) and Multivariate Regression Analysis, MRA (non-experimental design). GPS is particularly useful when there are few units or no subset of program non-participants (full coverage of the program) or their data are not accessible whereas program participants cover a wide range of treatments intensity (dose) or support levels. By contrast, MRA is particularly useful when there is a sample of program non-participants, but program participants are unique and thus it is challenging to identify a suitable comparison group.

Generalized propensity score matching (GPS) permits to estimate a dose-response model explaining treatment effect (response) depending on the level (dose) of treatment and other determinants. In this case, treatment regime or variable is not anymore a binary variable as in PSM, but instead a continuous variable. In other words, despite PSM explaining participation probability or treatment effect through a binary variable, Generalized PSM allows for multi-valued and/or continuous explained variable for treatment regime. As such, it is possible to estimate the status of a hypothetical control group by nullifying the dose or level of treatment or support.

Multivariate Regression Analysis, MRA controls for all relevant differences (pre-intervention and intervention or treatment related) between participating and non-participating groups affecting impact indicator (outcome variable). When relevant pre-intervention characteristics of two groups are captured in the observable covariates, MRA can yield an unbiased estimate of the intervention impact. The differences of the two groups (participants and non-participants) in terms of an impact indicator (e.g., conservation status for Natura 2000 sites), can be attributed either to the set of variables related to the intervention (treatment variables) or to others explaining the pre-intervention differences of them. In so doing, a hypothetical or non-experimental comparison group is constructed (nullifying pre-intervention differences) against which the net impact of a given intervention can be assessed. In the other words, MRA implies comparing participating sites (treatment group) with a non-experimental comparison group having no pre-intervention (treatment) difference with treatment group. However, estimated impact is valid only if it is possible to identify and then to take into account almost all possible differences between both groups (participating and non-participating) affecting impact indicator. In the other words, the more R^2 statistic of the estimated model tends to 1, the more the estimated impact will be reliable. However, as all program participants and non-participants must be compared across a high-dimensional set of pre-treatment characteristics, the method seems to be challenging.

3.2.2 ''Before-after'' approach

In certain cases such as full-coverage interventions (nationwide policies and programs), there is no scope for a comparison group (without or counterfactual situation) since the entire population participates. In such settings, there is almost no means to define or even to construct a control group. However, even in this case, change of the study scale and finding eventual non-participating sites at larger scale can be a solution. Otherwise, evaluating ''without'' situation as in ''with and without'' approach is no longer possible. The second best solution, i.e. ''before-after'' comparison remains thus the only means to measure net impact of a program (Lukesch and Schuh, 2010). ''Before-after'' comparison gauges impacts of an intervention using pre-intervention data of treatment group and compares them with the data collected for the same group after intervention. This method considers implicitly the before intervention or pre-Program data as a control group or ''without'' situation against which the impact of program will be gauged. However, the observed change, ignore the likely negative impacts of the absence of such intervention (e.g., a decline in biodiversity). In addition, the observed change might have occurred anyway (e.g., due to deadweight effect and general trends).

As such, the measurable change cannot be necessarily imputed to a given intervention since it can be due to certain self-motivated and/or program-independent actions of owners, called deadweight. Deadweight can be defined as a part of a change observed following a public intervention, which would have fully (or partially for partial deadweight) occurred, even without the intervention (DG AGRI, 2006b; Evalsed, 2012). For instance, certain forest managers may fully undertake measures in favour of biodiversity without supports of Natura 2000 network. Thus, the observed impact in the degree of biodiversity conservation cannot be fully imputed to this program. For instance, mid-term evaluation report of England rural development program (Elliott, 2010) revealed variable rates (from 19 to 50%) of deadweight effect for forestry measures (221, 223, 225, 227). Evidence on deadweight rate comes either from the surveys of beneficiaries and non-beneficiaries and/or from past research and evaluations. The estimated percentage of deadweight loss should be used to correct the estimate of program gross impact.

Similarly, the measurable change cannot be necessarily imputed to a given intervention since it can be due to other influences such as general or contextual trends, natural trends, external economic shocks and the impacts of other policy interventions etc. Thus, the observed gross change must be net out subtracting the impacts of other intervening factor especially that of ''deadweight'' effect and context related baseline indicators (e.g., general and/or natural trends).

The factors known as context related baseline indicators provide information on relevant aspects of general contextual trends (social, economic and environmental) that may have an impact on the performance of a public intervention. These indicators are useful especially when a before-after approach is used for impact assessment. As such, the trend of context related baseline indicators may help to explain their eventual contribution in the observed gross change after intervention and take them into account. For instance, an observed decrease in the content of nitrates and nitrites of a river (water pollution or impact indicator) can be caused not only by public interventions in upstream forests but also in large part by percentage of agricultural area applying agri-environment measures (e.g., decrease in fertilizer use). Thus, percentage of agricultural area applying agri-environment measures (at watershed or region level) can be considered as a context related baseline indicator (Elliott, 2010). Implementation of other programs or procedures such as ‘’sustainable forest management’’ or requirements of ecological labels (e.g., FSC) in Natura 2000 sites are another examples of context-related indicators. However, in many cases, although it is possible to assess impacts at micro level (e.g., at the level of intervention or site), it is difficult, or even meaningless, to relate this impact to general context-related trends (e.g., impact of other public interventions) which can be detectable rather at larger scales. For instance, as soon as the only public intervention at micro level (site level) is Natura 2000 program, impacts of context related indicators (general trends) on the conservation status of a habitat type at site level might remain often non-significant. However, the objective related indicator of a Natura 2000 site (e.g., conservation status) may evolve, even without any intervention, according to a natural trend (e.g., growth) at site level. As such, in the context of Natura 2000 program, the observed gross change should be net out taking into account not only ‘’contextual trends’’ but also ‘’natural trends’’. However, finding evidence on general trends at site level seems to be challenging, as natural trends at site level are often unknown, site specific and time variant. In order to mitigate the effect of general trends at site level, DID technique can be applied. Indeed, comparing before-after changes of program participating and non-participating sites, pre-intervention difference, common general trends and even deadweight effect can be net out. However, the crucial assumption justifying this method is that pre-intervention difference between participating (treatment group) and non-participating sites remains constant or time invariant, and this is not often the case. The ‘’before-after’’ approach especially for natural ecosystems such as Natura 2000 sites becomes thus challenging, complex and less reliable (Lukesch and Schuh, 2010).

3.2.3 Estimating net impact

Whatever the approach used (with and without or before-after), ‘’Net’’ impact or additionality corresponds to that part of the observed gross impact (difference in factual and counterfactual situations) that can be imputed to the intervention. Thus, net impact can be calculated once impacts of other intervening factors, especially direct effects such as ‘’double counting’’ and indirect effects like ‘’displacement’’, ‘’multiplier’’, ‘’substitution’’ and ‘’leverage’’ effects have been subtracted. Indeed, public interventions often carry certain ‘’leverage’’ and ‘’multiplier’’ effects and displace and/or substitute other activities. Unintended displacement (e.g., created jobs in an eligible area by displacing from or at the expense of another region) and multiplier effects (i.e., secondary and cumulative effects resulting from increased income and then consumption and repeating of the income-consumption cycle) can be considered as indirect effects of a public program. While these indirect effects especially on socio-economic variables (e.g., income and employment) sound often significant for certain region-specific public grants, their effects on environmental variables (e.g., biodiversity protection) under EU-wide Natura 2000 funding seems often irrelevant. Similarly, substitution effect (referring to a favorable change in the situation of a direct beneficiary at the expense of non-beneficiaries in the same area) and ‘’leverage’’ effect (additional private spending among program beneficiaries induced by public intervention but irrelevant to the intervention objective) on environmental variables sound irrelevant.

When before-after approach or DID estimator is to be used, deadweight effect should be also taken into account. As such, deadweight rate should be estimated surveying program beneficiaries. They were asked whether they would have proceeded with the program without its support or not. Deadweight corresponds to the percentage of respondents that would fully have proceeded with the project due to alternative funding sources or out of a desire to improve environment. The rest of respondents may proceed partially or may not implement the project at all without funding.

In addition, according to Elliott (2010) all programs needing public funding in order to get under way such as Natura 2000 (measure 224 under RDP) have generally low uptake because of a large part of ‘public good’ in the result. As such, this type of programs shows often less deadweight and the highest levels of additionality (i.e., low direct and indirect effects).

4. Assessment of public policy for biodiversity; methodological approach

4.1 Suitable impact indicator (response variable)

As the objective of the Natura 2000 network is to improve the conservation of biodiversity and habitats, objective related baseline or impact indicator can be defined using different indicators of biodiversity protection at the level of Natura 2000 sites. Thus, defining a relevant baseline indicator for biodiversity protection is of upmost interest especially for capturing impact of the Natura 2000 program on biodiversity protection.

It is noteworthy that given the huge diversity of life, biodiversity is an extraordinarily broad concept and, it is impossible to make rapid direct assessments of biodiversity in anything other than a superficial manner (Stork et al., 1997). However, biodiversity indicators, as surrogate measures of different aspects and components of biodiversity (Ferris and Humphrey, 1999) are crucial for studying and monitoring forest biodiversity as well as assessing impacts of management on biodiversity (Raulund-Rasmussen et al., 2011). Moreover, biodiversity indicators can be served to quantify and monitoring the state of biodiversity protection as a forest environmental service and finally, to estimate the cost of a given change in the quantity and/or quality of biodiversity protection. Most of biodiversity indicators have been proposed within the context of sustainable forest management at large scale (Ferris and Humphrey, 1999). However, several attempts have been made to define or to select indicators for forest managers to assess the effects of changing management practices on biodiversity at stand, forest or landscape levels (Ferris and Humphrey, 1999; Raulund-Rasmussen et al., 2011).

Generally, biodiversity indicators, as biodiversity components, can be divided in three groups: compositional (direct), structural (indirect), and functional indicators. Compositional indicators are dealing generally with species richness or the absolute or relative abundance of plant and animal species. However, it is extremely difficult and time/cost-consuming to assess directly biodiversity by enumerating all the organisms or even indicators species. By contrast, structural indicators measure the heterogeneity and the complexity of structure components (horizontal and vertical such as species, diameter or height diversity) of a habitat as an indirect measure of biodiversity. Indeed, habitat structure has significant impacts on biodiversity (via supporting a complex of habitat and niches enhancing diversity of associated species and its pivoting role in ecosystem functioning) and thus allows assessing the status of the latter indirectly (Ferris and Humphrey, 1999; Levrel, 2007). Being easily discernible and rapid to undertake, this approach plays a key role in monitoring of biodiversity (Ferris and Humphrey, 1999). In this category, deadwood quantity or quality as an individual component of the structure is a key indicator of biodiversity that is most-widely used structural indicators (Ferris and Humphrey, 1999). For Ferris and Humphrey (1999), indicators of biodiversity in managed forests have to be linked to forest type and management objectives. Functional indicators refer to ecological processes such as regeneration, decomposition, evapo-transpiration, nutrient cycling, gen flow etc. As such Functional indicators refer to ecological integrity (sustainability of ecosystem functions) while compositional and structural indicators represent the quality of biodiversity (Spanos and Feest, 2007). Functional indicators such as decomposition and evapo-transpiration are particularly valuable when assessing biodiversity in full sense (Spanos and Feest, 2007).

Most of biodiversity indicators have been proposed within the context of sustainable forest management at large scale (Ferris and Humphrey, 1999). The best relevant and practical indicators for forest management as for forest economics are those discernable by non-specialist and easy to assess. Moreover, they must be repeatable, cost effective and ecologically meaningful. From this point of view, certain structural indicators such as quantity and quality of deadwood or compositional ones such as tree species composition (e.g., extent and composition of broadleaved trees in coniferous stands) show generally a net advantage (Ferris and Humphrey, 1999). However, combining different aspects of biodiversity in a composite index (summative or synthetic indicators) such as conservation status sounds more comprehensive to simple indicators. Indeed, a simple indicator may reflect well one aspect of biodiversity (e.g., structure) and thus the impact of certain biodiversity actions or measures affecting directly this aspect (e.g., eliminating invasive species). By contrast, a composite index such as conservation status may serves as a unique impact indicator for a large set of biodiversity actions. Consequently, such an indicator may help enormously to analyze and compare simultaneously the impact of different biodiversity actions while this is not possible using a simple indicator of biodiversity (e.g., deadwood). Conservation status can be described as a situation where a habitat type or species is prospering (in both quality and extent/population) and with good prospects to do so in future as well (i.e., not only avoiding species extinctions but also providing excellent, good or medium protection). As such, this indicator represents at least two aspects of the biodiversity, its actual situation and its prospect. In addition, it combines different structural and functional indicators (e.g., deadwood, non-typical species, regeneration etc.). Moreover, this indicator corresponds to the quality of biodiversity protection (conservation) as the outstanding service of forests. Finally, favorable conservation status (FCS) is the overall objective to be reached for all habitat types and species of community importance (Natura 2000) and it is defined in Article 1 of the Habitats Directive. As impact indicator should be related to this overall objective, conservation status can be used to measure the extent to which this objective was reached. As such, positive changes in this variable may proxy improvements in the protection of biodiversity. These are why the degree of conservation has been selected as a suitable impact indicator for this research.

The Habitats Directive defines (according to criteria of Annex III) three classes of Conservation Status: A (excellent), B (good) and C (average or reduced) are describing the conservation status or 'the degree of conservation' of a habitat type or species in a specific site within former Standard Data Form, SDF. Status A and B are often considered as favorable status while status C (average or reduced) is referred to as unfavourable status. In the revised SDF (adopted in 2011) the term ‘conservation status’ at local level (site) is replaced by “degree of conservation” in order to reduce its confusion with ‘’Conservation Status’’ at regional level (Biogeographical and national assessments). According to Article 17 of the Habitat directive, the latter has to be done by all EU member States every six years from 2007 onward. For instance, recent evaluation of conservation status of all habitat type at national scale in France showed that only 17 percent of habitats are in favorable status (Status A and B) while 76 percent are in unfavourable situation (Bensettiti and Trouvilliez, 2009). Degree of conservation of a site explains generally its conservation status in terms of structure and functions of the natural habitat type concerned as well as its restoration possibilities.

As such, Article 17 of the Habitats Directive recommends employing ‘’the degree of conservation’’ as a baseline indicator for assessing the impact and efficiency of the Natura 2000 network. Consequently, this index can be also used in this research as an impact indicator for biodiversity protection. Similarly, degree of conservation of a habitat type at site level since the time of designation of a Natura 2000 site is considered in this research as a relevant objective related baseline indicator.

While pre-treatment and treatment data are often available either under standard data form or through payment agency, the data on response variable needs to be completed. Indeed, degree of conservation as a response variable is subject to first evaluation at designation phase of the site. The corresponding data can be obtained through either the standard data form or management plan. However, this data is qualitative and evaluated at site level. By contrast, a quantitative evaluation at intervention level will be needed for this study (Table 2). Although all EU member States must evaluate the conservation status of habitat types at biogeographical and national levels every six years from 2007 onward (Article 17 of the Habitat Directive), this is not the case at site level.

Table 2. List of response variables

Response variables	Time	Scale of data	Source
Degree of conservation	Pre-treatment	Habitat/Site	SDF/Management plan
Degree of conservation	Pre-treatment	Intervention parcel	Telephone survey/ existing data
Degree of conservation	Post-treatment	Intervention parcel	Sampling

4.2. Suitable pre-treatment and treatment (dose) variables

Dose-response models seem to be the relevant method for assessing biodiversity actions impact on unique Natura 2000 sites. The model relates response variable (outcome or impact indicator) to pre-treatment and treatment (dose) variables. As such, the intervention or treatment impact will be assessed using the coefficient of the treatment variables in the model. Estimating dose response model requires three groups of data on pre-treatment, treatment and response variables. As such, the major pre-treatment and treatment variables affecting response variable (degree of conservation of a habitat type at intervention level) for Natura 2000 sites (participating sites) must be indentified and collected.

Pre-intervention (pre-treatment) differences can affect directly or indirectly response variable (degree of conservation). Thus, to assess the net impact of conducted actions, the impact of pre-treatment differences must be brought out of the observed gross impact using major pre-treatment variables. For instance, degree of conservation of a habitat type targeted by a contract may well summarize its pre-intervention status and differences compared to other studied sites.

In order to identify major pre-treatment, treatment and response variables of technical, environmental and economic nature, all typical data of Natura 2000 sites within the standard data form, management plan and contract as well as the data related to financial data and implemented actions were reviewed. These data cover three distinct groups. First group includes characteristics of the Natura 2000 such as habitat type, area, forest cover, ownership, conservation status, number of species of Community importance, number of contracts etc. The second group covers existing tools of Natura 2000 network which are essentially management plan and signed contract. The third group includes different variable such as type of contract, duration of contract, grant amounts etc. The Tables 3 and 4 synthesize the most relevant covariates including pre-treatment and treatment variables. Although most of these data are at site level, they give the first idea about necessary data at intervention level.

Table 3. List of pre-treatment variables at site level

Type of variables	Pre-treatment variables	Description	Source
Site identification	**Site code**	The unique site code comprises 9 characters. The first two codes are the country code and the remaining 7 characters serve to create a unique alphanumeric code for each site.	**Standard Data Form of Natura 200 sites** (on website of European Environment Agency)
	Designated site name		
	Date of site designation	Ministerial order of designation	
	Type of site	Sites of Community Importance (SCI) and Special Protection Sites (SPA).	
Site location	**Site surface area**		
	Department	French department where the site is mainly located	
	Biogeographical region	where the site is mainly located	
Ecological information	**Degree of conservation for each habitat types**	First assessment for each habitat before site designation and contract signature	
	Number of forest habitat types		
	Code and Percent cover of habitat types on the site		
	Area of forest habitats		
	Number of Plants species justifying the interest of the site		
	Number of habitats of Community importance on the site		

Table 3. Continued.

Type of variables	Pre-treatment variables	Description	Source
Ecological information (continued)	**Number of forest habitats of Community importance on the site**		**Standard Data Form of Natura 200 sites**
	Total area of Site of Community Importance (SCI) in 2006 (ha)		
	Total area of Special Protection Sites (SPA) in 2006 (ha)		
	Total area of forest site of Community importance in 2006 (ha)		
	Total area of Special Protection Sites (SPA) in forest		
	Area of Natura 2000 sites in 2006		
	Area of forest Natura 2000 sites in 2006		
Site description	**Number of forest habitat classes**		
	Percent cover of habitat classes on the site		
	Site area belonging to public ownership		
	Site area belonging to private ownership		
	Share of public ownership in the site		

Table 3. Continued.

Type of variables	Pre-treatment variables	Description	Source
Impacts and activities in and around the site	**Number of human activities**	All human activities in and around the site affecting the conservation status and management of the site, either positively or negatively (listed in Appendix E).	
	Number of forest management activities	Number of forest management activities affecting the conservation status of the site	
	Influence of forest management activities	Impacts may be positive, negative or neutral	
	percentage of forest management activities whose influence is negative		
	percentage of forest management activities whose influence is positive		
	percentage of forest management activities whose influence is neutral		
	percentage of forest management activities whose influence is unknown		
Management plan	**The status of the management plan (NO doc, In preparation, Finished, approved)**		**French Ministry of Ecology**
	Approbation date of the management plan		

Table 4. List of treatment variables at site level

Treatment variables	Description	Source
Site code of contract	Site code where the contract was signed	**National agency for Payment, under Rural Development Program, European Commission**
Number of Natura 2000 forest contract		
The date of contract signature		
Start date for contract commitments		
The date of implementation of contract actions		
Legal form of the beneficiary of the contract		
Duration of contract		
Habitat or species aimed by contract		
Code of habitat or species aimed by contract		
Total cost of project borne by the contract	Total amount of grants received by the beneficiary during the period of the contract	
Annual amount of the contract	Total amount of annual grants received by the beneficiary of the contract	
Source of funding (EU, state, self-financing and public self-financing)		
Exemption from property tax (binary variable)		
Parcel code where actions is implemented		
Codes of conducted actions in the parcel	**Implementation cost of each unit of actions can be predictable or non predictable in advance**	
Physical unit of conducted actions in the parcel		
Costs of conducted actions		

Forest contracts finance non-productive investments aim at conserving or restoring forest habitats and /or associated species of community importance through certain managerial actions. In line with national and regional characteristics and requirements, these actions are often defined in Rural Development Program (RDP) of each European country. For instance, RDP of Hexagon, France (RDPH) conceives 13 actions (eligible for funding) for furthering biodiversity protection in Natura 2000 forest sites (MEDDTL, 2007). The physical unit (e.g., volume, number) or the cost of these actions can be also considered as treatment variables.

Table 5 presents possible effects of these actions on the core set of common criteria and indicators of biodiversity (habitat structure or functions) in forested Natura 2000 sites according to Carnino (2009a, 2009b).

Table 5. Possible effects of the selected and on-selected actions on the criteria and indicators of conservation status according to Carnino (2009)

Code	Actions	Directly affected criteria and indicators
F22701	**Creation or reestablishment of gap or shrub lands**	Severe impacts (eventual enhancing of invasive species) Typical flora
F22702	**Creation or reestablishment of forest ponds**	Not included in the Carnino methodology (it could be number of created microhabitats)
F22703	**Implementing managed regenerations**	Dynamics of regeneration
F22705	**Marking, felling or pruning without production objective**	Deadwood[1]/ density of standing dead trees and fallen deadwood Typical flora/percentage of non typical trees
F22706	**Maintenance and restoration of riparian forest and shore vegetation, and reasoned removal of deadwood**	Severe impacts Non-typical tree species
F22708	**Manual cleaning instead of chemical or mechanical ones**	Typical flora
F22709	**supplementary cost of Investment incurred by reducing the impact of road network**	Not included in the Carnino methodology
F22710	**Fencing habitat types of community importance**	Dynamics of regeneration
F22711	**Eliminating or limitation of invasive species**	Severe impacts/ percentage of invasive trees Deadwood/ density of standing dead trees and fallen deadwood

[1] Please note that the action effect is partially measurable with this indicator since according to the definition of deadwood, it takes into account only dead trees with DBH >35 and not all deadwood.

Table 5. Continued.

Code	Actions	Directly affected criteria and indicators
F22712	**Schemes enhancing senescent trees**	Very large living trees/density of large trees Deadwood/ density of standing dead trees and fallen deadwood
F22713	**Innovative operation in favour of species or habitats**	Not included in the Carnino methodology
F22714	**Investments for informing forest users**	Not included in the Carnino methodology
F22715	**Increase of irregular stand structure and texture for a non production objectives**	Not included in the Carnino methodology

Table 5 represents the relationship between treatment (i.e., actions enhancing biodiversity conservation) and response variables (i.e., degree of conservation). However, there are not any corresponding criteria and indicator in the methodology of Carnino for five actions. As such, they have a minimal effect or their impacts cannot be assessed through actual set of criteria in Carnino (2009a, 2009b). By contrast, Based on maximum score assigned to Carnino criteria, the most important (maximum score -60 and -20, respectively) as well as the easily and fully measurable criteria are directly related to the following actions: eliminating invasive species (action F22711), maintaining very large living trees (action F22712) and increasing deadwood (action F22705). As a result, selected actions strongly affect the Carnino criteria and indicators and thus the degree of conservation.

4.3 Suitable impact assessment method; Ordinal dose-response model

Suitable impact assessment method provides an reliable answer to the question if an intervention (e.g., Natura 2000 program) has brought forth intended net changes, which do not occur or would not have occurred without intervention (Lukesch and Schuh, 2010). In order to select a suitable method for Natura 2000 sites, method applicability and data availability as well as major specificities of Natura 2000 sites in terms of pre-treatment, treatment and response (impact indicator) variables affecting the choice of method have to be reviewed. The following sections review these aspects.

Method applicability and data availability

The inter-related issue of method applicability and data availability seems one of the most important factors determining the choice of assessment methods. Generally, there are two major approaches for impact assessment. Despite ‘’With and without’’ approach, assessing the net impact of an intervention through ‘’before-after’’ approach require estimating the effects of deadweight and general trends and subtracting them from the observed change. Thus, ‘’before after’’ approach is more complex, less reliable and more costly compared to ‘’with and without’’ approach. By contrast, ‘’with and without’’ methods seems more robust. They estimates directly the impact of an intervention through the simple difference between factual (with intervention or treatment group) and counterfactual (without intervention or comparison group) situations. However, ‘’with and without’’ approach implies that it is possible to select (experimental design), to find (Quasi-experimental design), to define (simplifying assumptions) or to construct (non-experimental design) a comparison group (see diagram1). In the case of Natura 2000 sites, the participating sites were selected for their habitats or species of community importance. As such, they may have different characteristics compared to non-Natura 2000 sites (non-participating sites). Moreover, even within Natura 2000 sites and in the same habitat type, great differences in terms of different characteristics can be seen (Demoly, 2010; Stenger et al., 2011). Finally, Natura 2000 sites as all complex ecosystems have often site-specific or unique characteristics (e.g., different potential and perspective even for the same degree of conservation). In such settings, selecting (experimental design), finding (Quasi-experimental design), defining (simplifying assumptions) or constructing (non-experimental design) a comparison group i.e., a set of non-Natura 2000 sites (non-participating sites) comparable to Natura 2000 sites (treatment group) having the same pre-treatment characteristics (including the degree of conservation) seems challenging. Even when identifying similar non-participating sites for studied Natura 2000 sites is feasible, their relevant data seems to be often less available compared to Natura 2000 sites. Indeed, it is quite costly to provide a comparable set of data for these non-Natura sites as we have for Natura 2000 site under Standard Data Form. By contrast, the data on Natura 2000 sites seems to be often available through different national or European online sources (Demoly, 2010; Stenger et al., 2011). Thus, application of all impact assessment methods based on both participating (Natura 2000) and non-participating sites seems challenging both in terms of method applicability and data availability within Natura 2000 sites. In this context, Bia and Mattei (2007) showed that regression based methods (non-experimental design) such as MRA can be also applied only for participating sites or enterprises (Natura 2000). As such, they applied MRA for estimating a dose-response model relating response variable (outcome or impact indicator) to pre-treatment and treatment (dose) variables. Thus, they inferred the intervention impact using the coefficient of the treatment variable in the model. Moreover, Bia and Mattei (2007) proposed a more complex method; Generalized Propensity Score Matching (GPS) based only on participating sites data. GPS methodology consists in estimating a dose-response model relating response or outcome variable (impact indicator) to the treatment and general propensity score as a measure of pre-treatment differences of treatment units. As explained by Bia and Mattei (2008), the implementation of the GPS matching method consists mainly of three steps:

-Identifying a transformation of the treatment variable, g(T), as a function of covariates (X), satisfying the normality assumption (Kolmogorov-Smirnov test) and the balancing property, respectively. Generalized Propensity Score (R) can be then estimated through the conditional distribution of the treatment variable (T) given the covariates (X): Ri = r(Ti, Xi).

-Estimating dose-response function i.e., the conditional expectation of the outcome variable Y, given R and T through an appropriate link function and estimator depending on the nature of the outcome variable (which may be binary, categorical, ordinal, or continuous)

-Averaging the estimated dose-response function over the score function (R) for each level of the treatment and plotting the estimated dose–response function.

Therefore, the estimation of the public intervention impact is based on the comparison of sites outcome with similar scores and different treatment (dose) level.

As a synthesis, it may be possible to assess net impact of Natura 2000 actions taking into account pre-treatment differences including site-specific ones and treatment differences of Natura 2000 sites through regression based or GPS methods. Bia and Mattei (2007) compared the estimated impact of public contributions to Piedmont enterprises on employment through both GPS and MRA. Considering the same covariates and range of treatment, the both methods provide coherent and similar results while MRA is much simpler.

The type of treatment and outcome variables is decisive for selecting the relevant assessment method. The next section deal with outcome variable and proposes appropriate methods given the type of these variables.

Continuous treatment variable (production method or technology)

The treatment can be described through either quantitative variables (measured in terms of numbers e.g., intensity of treatment) or qualitative variables (expressing a qualitative attribute e.g., having or not a treatment). When the intensity of treatment in terms of financial (e.g., public aides) or physical (e.g., chemical substances) quantities varies across the treatment units (or sites) as for Natura 2000 sites, the treatment variable is often measured using continuous quantitative variable. For instance, a set of variables including the level of annual financial support per hectare, type of conducted action and actions volume may describe treatment intensity. Most of the time, total financial support per hectare and per year (for maintenance or restoration of a Natura 2000 site) may proxy the treatment variable.

Ordinal outcome or response variable (impact indicator)

As reasoned earlier, degree of conservation was considered as a relevant impact indicator for evaluating Natura 2000 sites. Degree of conservation as all categorical (nominal) variables has no numerical meaning (Imai and Dyk, 2003). It is a discrete variable that takes on ordered values ranging from 0 to 100. As excellent degree of conservation (90-100) is better than good (70-89.9) and the latter is better than average (40-69.9) or reduced degree (0-39.9), the levels of the values can be naturally ordered. The term ordinal variable refers to such a discrete variable. Thus, the degree of conservation is a special type of categorical variables called ordinal variables. The type of outcome variable (quantitative or qualitative) affects the choice of assessment method. The Table 6 summarizes the compatible assessment methods given the type of outcome and treatment variables.

Table 6. The compatible assessment methods with respect to the type of outcome (or impact indicator) and treatment variables

Method	Type of response variable (impact indicator)	Type of Treatment regime
PSM	Qualitative	Qualitative
GPSM: linear dose-response function	Quantitative	Qualitative/ *Quantitative*
GPSM: Ordinal logit dose-response function	*Qualitative*	Qualitative/ *Quantitative*
Regression based model: Linear dose-response function	*Quantitative*	Qualitative/ *Quantitative*
Regression based model : Ordinal logit dose-response function	*Qualitative*	Qualitative/ *Quantitative*

As it can be seen in Table 6, impact assessment with a quantitative treatment variable and a qualitative outcome variable can be done through both GPSM and regression based model. In comparison to GPS, MRA seems to be more practical as the procedure of GPS is more complex, but it can produce similar results (Bia and Mattei, 2007). However, the both method can be used in the same time enabling a crosscheck of results. As for MRA and PSM, the extent to which the selection bias is reduced depends essentially on the richness and quality of the control variables or covariates (Bia and Mattei, 2007) describing pre-treatment differences of treatment units (e.g., Natura 2000 sites). This is why DID estimator is often recommended.

DID estimator of response variable

It is always recommended to combine DID (difference in difference) approach to GPS or regression based methods by using difference in the value of the response variable or impact indicator (i.e., degree of conservation) instead of its post-intervention value. The difference in fact should remove or mitigate selection error or “fixed effects” (constant over time) related to pre-treatment differences between sites. DID estimator can be directly applied to score-based ordinal response variable such as degree of conservation (see score range in Table 1). Moreover, it can be applied using a dummy variable describing the improvement or the reduction of the response variable against its no change (e.g., it is worth one for improvement and zero otherwise).

According to standard data form of Natura 2000 sites, most of data for all habitat types including response variable (the degree of conservation as an indicator of ecosystem health) are defined and provided at site level. Indeed, the degree of conservation of a habitat type at site level is a weighted average taking into account the status and the area of all parcel of this habitat type (including its areas under contracts) across the site. Thus, a relative large contract area under efficient actions may affect more the degree of conservation of the targeted habitat type at site level and vice versa. By contrast, the second evaluation of the response variable is accomplished naturally at contract level.

In order to apply DID estimator, the both evaluations of the response variable (impact indicator) must be obviously at the same scale. Being in the post-intervention phase, the re-evaluation of initial state at the same scale as the second evaluation is not possible. As such, only three solutions may be applied. The first solution is to upscale the second evaluation of the response variable from the contract level to the site level. However, such up-scaling seems to be extremely approximate. Moreover, the impact of the response variable at contract level on the weighted site level one is more likely negligible, especially when the contract area is small compared to the total area of the habitat throughout the site. Thus, up-scaling the response variable from contract level to site level seems inappropriate. By contrast, the second solution is to generalize the initial or the baseline evaluation of the response variable from the site level to the contract level. This solution is also extremely approximate as the conservation status of a habitat type at site level may have nothing to do with the status of the same habitat type at intervention level. The third solution is to estimate the initial state at contract level based on the existing information about pre-treatment state of studied contract area. Although obtaining relevant information on pre-treatment state of contract area is a challenging task, it seems the only satisfying solution. This is why this solution has been undertaken in this research. To do this, existing information on pre-treatment state of criteria and indicators of biodiversity conservation have been obtained via telephone survey or during field work.

4.4 Suitable scale and time span for impact assessment

Suitable scale

The question of scale is a key issue for assessing the impact of Natura 2000 actions on the status of biodiversity protection. The question of research scale should be dealt with considering ecological levels (habitat or species) and spatial scales (parcel, site, region etc.). From ecological point of view, there are two major levels; habitat type (ecosystem) and species. In other words, concepts, actions and indicators of biodiversity protection can be applied either for a species or for a habitat type (ecosystem). The loss of habitat is the main responsible of extinction and endangering fauna and flora species. All plants and wildlife depend on healthy ecosystem. In this sense, the level of habitat type was privileged in this research. Habitat types or species, in turn, can be assessed at different spatial scales including contract (parcel or polygon where intervention was done), site, biogeographical and national levels or even at larger scales.

Impact assessment should be done at contract or even intervention level (micro) before upscaling to higher levels (Lukesch and Schuh, 2010). Indeed, the micro level seems to be more feasible especially taking into account the multiplicity of intervening variables affecting biodiversity conservation (e.g., pre-treatment, context and treatment variables), which expand as the level goes up. Moreover, the micro-level analysis seems more reliable as the detailed and non-aggregated information on pre-treatment conditions, conducted actions, elapsed time and its new condition are available. By contrast, such information changing more or less from one parcel to another loses their precision and variability as the data are aggregated at higher scale. For instance, a good conservation status of a habitat type at site level may have nothing to do with the status of the same habitat type at contract level. Thus, larger scale may provide less information (due to aggregation) and higher number of covariates (due to higher scale). In addition, it may make no or little sense to assess the impact of a relative small amount of money (as the program participation is often low, it may be few thousand Euros within one or two contract per site) on a relative vast area of a habitat type over a Natura 2000 site or even at larger scale. However, large-scale assessments as within EU’s Rural Development Program or Habitat Directive present a whole program assessment at whole region, which are inevitable and quite useful for improving the efficiency of the program. The micro level assessments, in turn, help to better assess why and how a program has produced the observed effects. In this sense, conducting micro level assessments may be quite useful before applying larger scales assessments. This is why the research focuses on micro level assessment.

Natura 2000 payments in France are based on the contractual system. Every contract aims at a habitat type or a species. It plans and finances certain actions favouring the protection of biodiversity in the contract area. The impact of these payments can be logically assessed at contract or even intervention level, i.e., the parcel where an action has been conducted and financed. Contract or intervention level as the finest scale seems to be the most logical and relevant scale for assessing the impact and cost of biodiversity actions.

Suitable time span

Impacts of an action should be assessed at relevant time (after its implementation) since both environmental and socio-economic impacts of the action may emerge only after long time. Indeed, natural ecosystems often follow slow development paths (slowly unfolding changes) after an intervention and thus impacts of the intervention can be observed only after considerable time lags, especially in the case of biodiversity protection (Lukesch and Schuh, 2010). Although, full environmental impacts of an action can be assessed only after a long period, impacts begin to emerge after a minimum time span and then evolve over time (Kengen, 1997). Table 7 presents estimates of minimum time span for observing the first impacts of actions enhancing biodiversity.

Table 7. Expert knowledge estimates of minimum time span for observing the impact of actions enhancing biodiversity

Actions codes	Actions enhancing biodiversity	Minimum time to affect relevant indicators (years)
F22701	Creation or reestablishment of gap or shrub lands	5
F22702	Creation or reestablishment of forest ponds	-
F22703	Implementing managed regenerations	1
F22705	Marking, felling or pruning without production objective	1
F22706	Maintenance and restoration of riparian forest and shore vegetation, and reasoned removal of deadwood	5
F22708	Manual cleaning instead of chemical or mechanical ones	5
F22709	supplementary cost of Investment incurred by reducing the impact of road network	-
F22710	Fencing habitat types of community importance	5
F22711	Eliminating or limitation of invasive species	1
F22712	Schemes enhancing senescent (large) trees	5-10 years for Large trees For deadwood the time varies according to the relation between mortality rate and DBH of large trees
F22713	Innovative operation in favour of species or habitats	-
F22714	Investments for informing forest users	-
F22715	Increase of irregular stand structure and texture for non -production objectives	-

As it can be seen in the Table 7, the minimum time span for observing the first impacts of selected actions (F22705, F22711, F22712) on relevant indicators varies logically from 1 year (F22705 and F22711) to more than 5 years (F22712). It is noteworthy that the action F22712 enhancing senescent trees may affect not only the number of very large trees per hectare but also number of dead trees per hectare. Indeed, according to the relation between mortality rate and DBH of large trees, schemes enhancing senescent (large) trees may raise the number of dead trees. Mortality rate-size relationship may be decreasing, ladle-shaped or constant, which varies across species and sites (Shimatani et al., 2008). However, ladle-shaped relation and specially increasing mortality rates has been often observed for large, old-age trees. The latter is may be due to the loss of vigour, entering into old and decay phase and higher susceptibility to storm and pests (Wunder et al., 2008). As such, the minimum time to affect deadwood indicator varies depending on mortality rate-size (DBH) relationship and the number of senescent (large) trees per hectare.

Within Habitat Directive, a time span of 6 years is advised for the assessment and reporting of conservation status of all habitat types (Evans and Arvela, 2011). This time span can be used to make a first assessment of an intervention impacts. However, a full assessment of impacts and its pattern over time may be feasible while long time series of baseline and impact indicators and interventions schedule are available. First implementation of the Natura 2000 program in member State began in 1997-8 (Rekola, 2003; Marage and Delmas, 2008). In France, the first financial data on Natura 2000 contracts covers the period of 2002-2007. As such, financial data of contracts in the period 2002-2007 has been privileged for this study instead of more recent data of the period 2007-2010. In so doing, we conceive a time span between 5 to 10 years (average of 7 years) between first evaluations of conservation status within FSD, actions implementing and second evaluation. However, only a first assessment of Natura 2000 impacts seems actually to be possible. By including the following years, the stability of estimated impacts can be re-evaluated over time.

4.5 Evaluating the degree of conservation before and after intervention (response variable)

A qualitative method based on expert opinion has been often used for assessing the degree of conservation. However, a new quantitative method based on a scoring system has been recently proposed and applied for forest habitats (Carnino, 2009a, 2009b) at both polygon and site level. The scoring system relies on three categories of criteria; structure, function as well as pressures and threats or restoration possibilities (Carnino, 2009a, 2009b; Evans and Arvela, 2012). For each category, there are several criteria and indicators associated each one to a score. If the value of a criterion corresponds to its favorable range, then the score will be zero. Otherwise, the criterion takes progressively negative score depending on the extent of its deviation from favorable range (Carnino, 2009a, 2009b). Criteria and indicators describing the structure and function of a forested habitat includes non-typical tree species, severe impacts, very large trees, dynamics of regeneration, deadwood, typical flora, scatter impacts. These criteria and associated indicators (measured variable) have been synthesized in Table 8.

Table 8. Favorable and unfavourable ranges of structure and function indicators and corresponding scores according to Carnino (2009)

Criteria (structure and function)	Indicators at site level	Possible scenarios	Score penalty
Non-typical tree species	**Percentage of basal area of non-typical tree species**	None of them	0
		1-5% non-typical sp. and any invasions	-5
		5-15% non-typical sp. or <15% invasions	-10
Severe impacts **(e.g., invasive species, soil erosion, hydrological disturbance)**	**e.g., Percentage of basal area of invasive species**	15-30% non-typical sp. or 15-30% invasions	-30
		>30% non-typical sp. or >30% invasions	-60
Very large living trees (senescent, large or big trees[2]) **(According French institute of forest inventory, i.e., all trees with DBH > 45**	**Number of stem per hectare of very large trees**	Number of very large trees per hectare ≥ 5	0
		Number of very large trees per hectare 3-5	-2
		Number of very large trees per hectare 1-3	-10
		Number of very large trees per hectare < 1	-20
Dynamics of regeneration	**Area of young stands** **(even-aged and coppice stands)**	Area of young stands 5-30%	0
		Area of young stands 5%> 30%<	-10
	Regeneration problems **(uneven-aged stands)**	Any regeneration problem	0
		Regeneration problem	-10
Deadwood **(fallen or standing)**	**Number of dead trees having DBH ≥ 35 per hectare**	Number of deadwood 6 <	0
		Number of deadwood 3-6	-2
		Number of deadwood 1-3	-10
		Number of deadwood < 1	-20
Typical flora	**Percentage of typical species**	Percentage of typical species 40%<	0
		Percentage of typical species 20%-40%	-5
		Percentage of typical species 20%>	-10
scattered impacts **(e.g., fires, browsing pressure, human overcrowding)**	**Observed major damages** **(e.g., bite damage)**	Negligible or no damage	0

At first, a starting score of 100 (Excellent, A) is assigned to all habitat type across the site. A final score is then calculated for a habitat type at site level according to the average value for each criterion across the site (different polygons of the habitat type). When the average value of

[2] According to Angelstam and Dönz-Breuss (2004) all trees with DBH superior to 80 cm can be considered big or large trees (the cited reference is quoted in Cantarello and Newton, 2008). However, According French institute of forest inventory, all trees with DBH>65 for broadleaves and>70 for conifers can be considered as large trees. In practice, a threshold of 45-50 cm of DBH is used to mark a tree for biodiversity objectives and to calculate the necessary compensation for maintaining a biodiversity tree since this threshold corresponds in average to target diameter. As such, the Carnino method should precise exactly the definition of very large trees.

all criteria is in optimal condition, the degree of conservation remain 100 (habitat has typical species, balanced age-structure and a significant amount of dead wood). Otherwise, the starting score decreases by the sum of negative scores of non-optimal criteria. This gives a final score out of 100 for a given habitat. This score is finally compared to the "threshold values" to assess the degree of conservation (Carnino, 2009a, 2009b).

Table 9. Score range and corresponding degree of conservation according to Carnino (2009 a, b)

Score range	Degree of conservation		Status class
0-39.9	Reduced	C	Unfavourable
40-69.9	Average	C	
70-89.9	Good	B	Favourable
90-100	Excellent	A	

As it can be shown in the Table 9, the degree of conservation is one of several possible categories: A (excellent), B (good) and C (average or reduced). Table 10 presents typical values of the criteria and indicators determining degree of conservation.

Table 10. The criteria and indicators determining degree of conservation according to Carnino (2009) and their typical values for degrees of conservation A and B

Criteria (structure and function)	Indicators at site level	Degree A (Excellent)	Degree B (Good)	Global Score
Non-typical tree species	**Percentage of crown cover**	0%	15<%	≥ 70
Severe invasions **(e.g., invasive species, soil erosion)**	**Percentage of crown cover**			
Very large trees **(According French institute of forest inventory, i.e., all trees with DBH>65 for broadleaves and>70 for conifers)**	**Number of stem per hectare**	≥ 5	≥ 3	
Dynamics of regeneration	**Area of young regeneration** **(for even-aged stands)**	5-30%	5-30%	
	Regeneration problems	0	0	
Deadwood	**Number of dead trees having DBH ≥ 35 per hectare**	≥ 7	≥ 3	

Table 10. Continued.

Criteria (structure and function)	Indicators at site level	Degree A (Excellent)	Degree B (Good)	Global Score
Typical flora	**Percentage of typical species**	≥ 40%	≥ 40%	
scatter invasions (e.g., fires)	**Observed major damages**	Null or negligible	?	

As it can be seen in the table, the criteria and indicators of the degree of conservation allow not only to combine different aspects of biodiversity in a single composite index but also to propose appropriate actions (e.g., eliminating invasive or non-typical species) to enhance the conservation status of the habitat. Moreover, an aggregation of the results of the conservation status assessments on local sites could directly give a result on biogeographical level according to the requirement of to article 17 of the Habitats Directive (Evans and Arvela, 2011). Degree of conservation can change or remain unchanged according to the time and measures required to restore many habitat types and species to recover from unfavourable status.

5. Estimating provision cost function of biodiversity protection

5.1 Definition, components and methods of estimation

As for values, cost of provision for an environmental service can be defined as the net or additional costs (per hectare per year) of a given change in the quantity and/or quality of a service through doing certain favorable managerial actions (treatment or action cost) furthering the service and/or not doing certain unfavourable actions and activities (opportunity cost). For biodiversity protection, provision cost can be defined as the net cost of a given net improvement in a relevant indicator explaining the quality of protection. If the indicator of the service such as the quality of protection is a continuous variable, provision cost can be also defined as a unit cost of provision explaining the provision cost of each unit of the service quantity (Richards and Stokes, 2004). As such, provision cost is a function of the net impact of conducted actions on biodiversity conservation.

Compensation payments for furthering the conservation of biodiversity within Natura 2000 sites rely mainly on input or actions costs (e.g., in France) as well as on opportunity costs (e.g., in Denmark, Latvia). For instance, within Natura 2000 measure of RDP in Latvia, compensation payment is estimated equal to 120 €/ha/y if all forestry activities is banned, 80 €/ha/y if only final felling is forbidden and 40 €/ha/y if the clear-cut is prohibited. Thus, total payment within Natura 2000 contracts can be expressed by equation 1:

$$TP = \mathrm{MC} + \mathrm{RC} + \mathrm{OC} \tag{1}$$

TP: Total payment or total amount of forest contract

MC: Maintenance actions cost

RC: Restoration actions cost

OC: Opportunity cost (e.g., land cost)

Unlike total payment (TP) which is a function of inputs (actions) quantities and prices, provision cost as production cost is a function of output quantity or quality (e.g., goods or service). As such, total payment for a given net change in the quantity or quality of biodiversity conservation can be considered as provision cost of biodiversity conservation. In general, provision cost is a function of initial status of forest (and thus the necessary quantity of actions), opportunity cost and external effects outside the treatment area (Richards and Stokes, 2004) as well as the net change in the output. For the same input costs, the provision costs could be low if the efficiency of conducted actions on the output is high and vice versa.

The question of provision cost of forest services has been rarely dealt with in the literature. Moreover, most of these few researches focus on carbon sequestration while provision costs of other services such as biodiversity protection has not received enough attention. These researches were often done to evaluate the cost-effectiveness of the forestry option as an instrument (carbon sink) for atmospheric carbon dioxide and climate change mitigation. All these research identify at the outset forestry practices to increase a forest service like carbon sequestration, i.e., the actions generating variable costs.

The optimal provision of the majority of the forest goods and services may require specific targeted or service-specific actions, which generates variable cost (treatment or action costs), being explicit or implicit, and thus implies investments. These variable costs benefiting directly the targeted service can be considered as direct costs as well. These costs have been acknowledged and cited in the Guidance Handbook for Financing Natura 2000 of the European Commission as ‘’ongoing habitat management and monitoring costs’’ (Miller et al., 2006). For instance, Richards and Stokes (2004) review nine forestry practices and actions employed to increase carbon sequestration on forestland (e.g., modifying forest management). Another example is the Rural Development Program of Hexagon, France (RDPH), which propose thirteen actions to further biodiversity protection in Natura 2000 forest sites. Table 11 present these actions (in its original language, French and in English), their type and their impact mechanism on biodiversity.

Table 11. List of 13 eligible biodiversity actions for furthering biodiversity protection

Code RDPH and name of actions in		Type of actions	Impact mechanism on biodiversity
French	**English**		
F22701 - Création ou rétablissement de clairières ou de landes	Creation or reestablishment of gap or shrub lands	Restoration	Creation of micro habitat
F22702 - Création ou rétablissement de mares forestières	Creation or reestablishment of forest ponds	Restoration	Creation of micro habitat
F22703 - Mise en œuvre de régénérations dirigées	Implementing managed regenerations	Maintenance	Regenerations favouring
F22705 - Travaux de marquage, d'abattage ou de taille sans enjeu de production	Marking, felling or pruning without production objective	Restoration	Creating spaces in favor of animal and/or plant species of community importance
F22706 - Chantier d'entretien et de restauration des ripisylves, de la végétation des berges et enlèvement raisonné des embâcles	Maintenance and restoration of riparian forest and shore vegetation, and reasoned removal of deadwood	Restoration	Preservation of shore habitat with plantation
F22708 - Réalisation de dégagements ou débroussaillements manuels à la place de dégagements ou débroussaillements chimiques ou mécaniques	Manual cleaning instead of chemical or mechanical ones	Maintenance	Preservation of soil fauna and flora
F22709 - Prise en charge de certains surcoûts d'investissement visant à réduire l'impact des dessertes en forêt	supplementary cost of Investment incurred by reducing the impact of road network	Maintenance and restoration	Reducing negative impacts of roads traffic and technical design on forest biodiversity
F22710 - Mise en défense de types d'habitat d'intérêt communautaire	Fencing habitat types of community importance	Restoration	
F22711 - Chantiers d'élimination ou de limitation d'une espèce indésirable	Eliminating or limitation of invasive species	Restoration	Favouring native and typical species
F22712 - Dispositif favorisant le développement de bois sénescents	Schemes enhancing senescent trees	Maintenance	Maintaining the number of very large trees functioning as habitat trees
F22713 - Opérations innovantes au profit d'espèces ou d'habitats	Innovative operation in favour of species or habitats	Restoration	Innovative actions such as creation of forest edge or ecotone
F22714 - Investissements visant à informer les usagers de la forêt	Investments for informing forest users	Maintenance	Forest extension locally or at larger scale
F22715 - Travaux d'irrégularisation de peuplements forestiers selon une logique non productive	Increase of irregular stand structure and texture for a non production objectives	Restoration	

Explicit or implicit costs of these actions are often the first type of provision costs that are typically acknowledged by land-owners before entering into a legally binding contract for providing environmental services under Natura2000 network, EU's rural development program or similar programs/funds (NEWFOREX, 2009). These costs may be divided into maintenance and rehabilitation/restoration costs depending on the type of conducted actions as they are presented in the Table 11.

Restoration actions involve focused activities with specific objectives to reach a specific habitat state. According to the definition of Society of Ecological Restoration (Society for Ecological Restoration International Science and Policy Working Group, 2004), restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. The restoration involves the re-establishment of the pre-existing biotic integrity in terms of species composition and community structure, i.e., restoring or bringing back to original, healthy and vigorous state. By contrast, maintenance actions call for variable treatments within the normal ranges often for a restored ecosystem or an already healthy ecosystem over time in order to preserve habitat conditions, species composition and community structure. Once an ecosystem is restored or as soon as it is in its original, healthy and vigorous state, ecological processes ensure self-maintenance in the ecosystem. As such, the maintenance cost is often conceived to be much lower than costly restoration actions. However, the maintenance cost of forest structure such as those of very large trees is not necessarily negligible as the opportunity costs (implicit costs) of immobilizing these financially mature trees and their occupied soil seems to be high, especially for a large numbers of trees over long time span.

Restoration actions generate initial treatment costs, which are generally expressed as capital outlays since they benefit the protection of biodiversity for more than one year. For instance, actions such as creation or reestablishment of water bodies and standing deadwood in forest aimed at enhancing the protection of biodiversity through reconstruction and/or improvement of certain habitats. The costs of these actions, capital outlays, benefit the protection of biodiversity for more than one year. Similarly, all maintenance costs involving multiple years actions (e.g., Schemes enhancing senescent trees) benefit the protection of biodiversity for more than one year. As such, in order to have an annual basis for all costs, these costs should be converted to an equivalent annual cost given the useful life of capital assets or project or contract duration and real discount rate. As such, all costs should be expressed per year per hectare.

According to objective and data availability, cost of biodiversity protection can be estimated based either on the type of conducted actions (e.g., restoration or maintenance actions) or on all actions taken. However, the treatment costs corresponding to these two groups of actions, i.e., maintenance and restoration should be distinguished. Indeed, treatment costs depend largely on the initial status of forest. As such, provision cost of biodiversity protection could be low if treatment costs cover only maintenance actions. By contrast, it could be higher if restoration actions are required. As such, treatment costs should be divided into maintenance and restoration costs based on the type of conducted action.

Besides the explicit variable costs of these specific targeted actions, the provider of the service may experience certain implicit costs. In order to maintain or enhance biodiversity, forest owners may be often obliged to fully or partially deviate from the optimal management of the forest for wood production or other economic activities. This deviation generates certain implicit costs. Thus, the provision cost is also related to the definition of the baseline scenario i.e., standard or optimal management, which is often based on wood production but could rely on other jointly produced goods and services as well. Indeed, the costs and impacts of a forest service-enhancing program such as Natura 2000 must be evaluated relative to what would have arisen in the absence of such a program, referred to as the baseline scenario. As such, the provision of a service like biodiversity protection may cause a decrease in the quantity and/or quality of certain market goods and services (or even non-market) such as wood production (e.g., harvest ban) compared to the baseline scenario. For instance, the maintenance of forest structure such as those of very large trees imposes certain implicit costs (opportunity costs) due to immobilization of these financially mature trees and their occupied soil over time.

In such cases, provision cost should also include the opportunity cost related to the partial or full loss (land cost) of wood revenue. By contrast, the provision cost of biodiversity protection may bring forth not only a given improvement in the status of biodiversity protection but also certain secondary improvements in the quantity and /or quality of other jointly produced services (Gantioler et al., 2010). Many examples exist for this situation, from more known services such as water regulation and water purifying to provision of pollinators and natural enemies of pests. In such settings, all conducted costs for biodiversity protection should be imputed not only to biodiversity protection but also to all other enhanced services. As a result, provision cost of biodiversity conservation may be considered not only as the cost of conservation but also as the cost of maintenance or restoration of all services provided by biodiversity. In addition, this implies that provision costs need to be assessed in an integrated way taking into account not only related market goods such as wood but also other related non-market goods and services in the same time (NEWFOREX, 2009). However, it is a fully complex task to assess the impact of a program such as Natura 2000 on all forest goods and services, especially when the relevant data is lacking. This is may be why the most of studies assessing provision cost focus on a single good or service (e.g., carbon sequestration) relative to a baseline scenario based mostly on market goods and services such as wood. The latter is more tangible especially for private sector since its data is more accessible (through standard forest inventory data) compared to non-market services. In addition, variable costs may include transactions costs and risks (within long-term contracts excluding future options) of entering into binding contracts for the provision of a forest service. However, 5 years-contracts of Natura 2000 in France may generate few risks related to exclusion of future options compared to long-term contracts. As such, at micro level and from provider perspective, provision cost includes implicit (treatment or actions costs) and explicit costs (e.g., opportunity partial or full land cost i.e., loss of land revenue deviating from the baseline scenario) as well as variable transactions costs and risks.

Provision cost of biodiversity protection; components

The Guidance Handbook for Financing Natura 2000 of the European Commission define and classify all relevant costs of biodiversity conservation within Natura 2000 network (Miller et al., 2006). These costs are divided into four categories: establishment of Natura 2000 sites, management planning, ongoing habitat management and monitoring, and investment costs. There are a total of 25 activities eligible for funding amongst these four categories (Table 12).

Table 12. Components of provision cost of biodiversity conservation according to the Guidance Handbook for Financing Natura 2000 (Miller et al., 2006)

	Types of Activities
Establishment Natura sites	1. Administration of site selection process
	2. Scientific studies/inventories
	3. Preparation of initial information
	4. Pilot projects
Management planning	5. Preparation of management plans
	6. Establishment of management bodies
	7. Consultation – public meetings
	8. Review of management plans
	9. Running costs of management bodies
	10. Maintenance of facilities for public access
	11. Staff
Ongoing habitat management and monitoring	12. Conservation measures –habitats
	13. Conservation measures –species
	14. Conservation measures:
	15. Agreements with owners and managers
	16. Provision of services; income loss; etc
	17. Monitoring and surveying
	18. Risk management
	19. Surveillance of the sites
	20. Provision of information/publicity material
	21. Training and education
	22. Facilities to encourage visitor use
Investment costs	23. Land purchase and others
	24. Infrastructure for restoration

Amongst these four major components of provision costs, only the third components; ongoing habitat management and monitoring, includes measure or actions enhancing biodiversity. These costs may be conceived as variable costs as they affect directly the status of biodiversity as the principal output and objective of the Natura 2000 network. By contrast, other components i.e., establishment of Natura 2000 sites, management planning and investment costs provide and maintain the Natura 2000 network and its structure and infrastructures (e.g., management bodies and plans) support the Natura 2000 sites among others to meet requirements of EU funds (e.g., to prepare management plans, to inform and convince owner to enter in a contract etc.). As such, these costs (e.g., preparing and revising management plans or management plans and establishing and maintaining management bodies at site level) may be considered as fixed costs since they don't affect directly the status of biodiversity as the principal output and objective of the Natura 2000 network. This is why the annual amount of these costs per hectare do not often vary across the Natura 2000 sites and could be easily estimated (Hermoso Barroso, 2012). By contrast, variable costs depend on the type and the quantity of conducted actions and vary highly across Natura 2000 sites. Moreover, the total amounts of Natura 2000 contracts financed under EU's Rural Development program (RDP) or other funds are calculated based on the estimated explicit and implicit costs of recommended actions (variable costs). As such, this research deals with forest contracts, it focuses only on the variables costs of biodiversity protection.

Scale, perspective and method of cost evaluation

The extent of provision costs of forest services vary with spatial scale and evaluation perspective. For instance, opportunity cost from private perspective at small scale may correspond only to the partial or full loss (land cost) of wood revenue while it may generate higher costs at larger scale including the raise in wood price and land costs. The opportunity cost from social perspective may be still higher including the raise in agricultural products prices. From society perspective, external effects outside the treatment area (e.g., economic transboundary or side effects in other connected sectors and/or land uses) seem to be also important. However, identifying the social costs associated with conversion of biodiversity seems to be a particularly difficult task.

Richards and Stokes (2004) identified three general methods to evaluate land cost or more generally provision cost of a service: bottom-up engineering cost studies, sectoral optimisation studies and, more recently, econometric studies of the revealed preferences of land-owners. Bottom-up method estimates exogenously the additional incurred cost of providing a service using the value of all necessary inputs (e.g., actions) ranging from explicit cost of actions to implicit or opportunity costs. By contrast, the two last methods estimate land cost based on land use conversion analysis (e.g., agriculture and forests) in response to differences in prices of relevant products (e.g., wood and crop). The second method estimate cost based on the optimization of landowner's welfare in all related sector over time while the third one relies on revealed preferences through historical data. Land cost is estimated then considering changes in land uses and production at regional scale over a time period compared to baseline scenario.The two last methods apply at county or regional level and could be more efficient than the first method but it is costly and difficult to apply. By contrast, the first method applies at different scales especially at small scale. It is relatively transparent and simple to interpret. This is why it is the most widely used among the three methods. However, it includes almost no consideration of behavioural responses of landowners, related markets or other economic actors (e.g., increasing marginal costs of land as a hypothetical carbon sequestration program expands). Thus, its application may be more pertinent at small scale rather than large scale. Given the low uptake of Natura 2000 program and the scale of the analysis at contract level (see section 3.6), the bottom-up engineering cost models seems to be pertinent. In addition, this method corresponds well to the nature of financial data within Natura 2000 contracts. As such, this research deals with provision costs of biodiversity protection at micro level and from provider perspective based on total amount of forest contracts estimated according to bottom-up engineering cost studies.

5.2 Provision cost function

Let's assume a forest owner who produces a vector of outputs (in short term) or impacts (in long term) $y \geq 0$ (see output/impact definition in the next section). The production process generally uses natural and economic capitals (K), labour (L), energy (E), material (M) and the production process can be represented as f (y, x, T; Z) = 0, where x = (xK, xL , xE , xM) is the vector of inputs, T is the type of conducted action and Z represents environmental variables having an impact on costs.

The duality result of production theory is specifying a provision cost function that is dual to the production function. Given the vector of (positive) input prices w ≡ (wK,wL,wE,wM) > 0, we assume that the forest owner chooses its inputs so as to minimize long-run cost of production:

$$min_{x\geq 0} \sum_{i=K,L,E,M} w_i x_i \tag{2}$$

Subject to f (y, T, x; Z)

This yields the long-run provision cost function

$$C_{LR}(y,T,w;\ Z) = \sum_i \mathrm{w_i}\ \mathrm{x_i}(\mathrm{y},\mathrm{T};\ \mathrm{Z}) \tag{3}$$

The long-run cost function has to verify the following properties:

• $C_{LR}(y,T,w;\ Z)$ is non negative and non decreasing in $y \geq 0$ and $w > 0$;

• $C_{LR}(y,T,w;\ Z)$ is homogenous of degree 1, concave (which rounds inward) and continuous with respect to w.

The provision cost function,$\mathrm{C_{LR}}$, is a function of output or impact and all environmental variables like initial conservation status. As such, in order to estimate provision costs (additional cost, per hectare per year, of a given change in the quality of biodiversity conservation), the output or the impact of investments should be assessed. The latter will be dealt with in the next section.

Applying DID estimator on response variable (output or impact), the observed change in the degree of conservation, yi-yo , can be specified as a function of total provision cost or total payment (CLR), type of treatment action (T) and pre-treatment environmental variable, Z.

$$Y_i\text{-}Y_0 = f(C_{LR}, T;\ Z) \tag{4}$$

As it was discussed above, Yi-Yo is an ordinal variable and thus this ordinal logit model should be estimated using maximum likelihood estimator. By contrast, provision cost, C_{LR}, assumed to be an exogenous variable. However, C_{LR} is an endogenous variable as it is also a function of yi-yo

$$C_{LR} = f(y_i\text{-}y_{o,}\ T_,\ Z) \tag{5}$$

As such, estimation of equation 8 seems to be complex since the problem of ordinal dependent variable and endogeneity of independent variable must be dealt with simultaneously. By contrast, it is easier to estimate equation (5) since its dependent variable is continuous and only the endogeneity problem should be solved. For a regression equation with endogenous variables and heteroscedastic errors, the generalized method of moments (GMM) would seem to be the best method for convergent and efficient estimations of the parameters.

The basic idea underlying estimation by GMM consists of specifying moments without specifying the parametric form, in order to construct conditions of orthogonality that will be used to identify the equation. The equation can be written in the following form:

$$Y = R\beta + \varepsilon \tag{6}$$

Where Y is the dependent variable, R is the regressor (i.e., explanatory variables) matrix, β is the associated parameter vector and ε is the error vector.

By using the expression W to represent the instrument vector, all conditions of orthogonality (or of moment) can be written as follows:

$$E(\varepsilon W) = 0 \tag{7}$$

In practice, these conditions are approached by their empirical equivalent that is a positive semi-defined weighted matrix, enables us to construct the GMM criterion to be minimized in order to obtain the estimator for β. The GMM criterion builds upon the empirical counterpart by considering the following quadratic form:

$$\left(\frac{1}{N}W'(Y - R\beta)\right)' \Omega^{-1} \left(\frac{1}{N}W'(Y - R\beta)\right) \tag{8}$$

Where N is the number of observations, $\Omega = E[W'(Y - R\beta)(Y - R\beta)'W]$ is the L×L variance-covariance matrix of the set of orthogonality conditions.

GMM estimation typically proceeds in two steps. This consists first of estimating the equation with 2SLS (two-step least squares) method to construct the weighted matrix from the residuals, and then minimizing the GMM criterion constructed from the weighted matrix calculated in the first step. The GMM estimator based on this method is robust for all forms of heteroscedasticity and efficient. It is:

$$\hat{\beta} = \left(R'W\hat{\Omega}^{-1}W'R\right)^{-1}\left(R'W\hat{\Omega}^{-1}W'Y\right) \tag{9}$$

Where $\hat{\Omega} = \frac{1}{N}\sum_{n=1}^{N} W_n'\hat{\varepsilon}\hat{\varepsilon}'W_n$, with $\hat{\varepsilon}$ the first-step 2SLS residual.

Once the parameters have been estimated, it is then necessary to ensure that the instruments chosen verify the moment conditions constructed on the basis of the hypotheses produced by the econometrician. To do this, a test is carried out on the overidentifying constraints, based directly on the GMM criterion. If L expresses the number of moment conditions (corresponding to the number of instruments multiplied by the number of equations) and K the total number of parameters to be estimated, the criterion follows a χ^2 distribution with $L-K$ degrees of freedom under the null hypothesis for validity of the moment conditions. This is known as the Hansen specification test for which there are $L-K$ over identifying constraints for which it is not necessary to identify the parameters but which transmit information about the model specification. In particular, if the instruments contain variables that are not exogenous, the Hansen test will detect them and the whole set will then have to be modified.

6. Data collection and processing

6.1 Data collection

After preparing detailed field work map and program via telephone survey, all relevant information on pre-treatment and post-treatment status of biodiversity indicators has been measured using two major sources and different methods.

6.1.1 Existing data set (pre-intervention)

EMIN2K is the acronym of a research project financed by the French Ministry of Ecology, Sustainable Development, Transport and Housing and conducted jointly by the Laboratory of Forest Economics (LEF) and the Laboratory of forest-wood resources (LERFOB), mixed research units of INRA/AgroParisTech. The database EMIN2K has been constructed using two main data sources. The first source is the Natura 2000 database, which is freely available online either on the website of French institution called National Inventory of Natural Heritage

(INPN) or on the website of the European Environment Agency. This database contains all information of Standard Data Form (SDF) of the Natura 2000 sites. The second source includes financial and physical data of Natura 2000 contracts recorded and provided by service and payment agencies in all EU State members within the Rural Development Program. EMIN2K data covers two periods, the first one from 2002 to 2007 and the second one from 2007 to 2010. The first period include 375 forest contracts over 97 forested Natura 2000 sites (containing at least one forest habitat type) for which all SDF information are available. The construction of this database within the EMIN2K project lasted 1 year. The use of this database for this project has been authorized under a convention with the Laboratory of Forest Economics. However, this information covers only pre-treatment or pre-intervention state of sites while the data on post-treatment or post-intervention states is lacking. Nevertheless, estimation of provision costs relies on post-intervention data (for assessing the impact of conducted actions). As a result, this database had to be completed with telephone survey and field inventory. A summary of existing data on pre-intervention state of studied intervention parcels has been synthesised in Table 13.

Table 13. Summary of existing data on pre-intervention state of studied intervention parcels

N	Site code	Forest contract/parcel	Habitat type code	Action code	Action cost €/ha/y
1	FR8212006	073NA050003 P1	H9130	F227005	400
2	FR2100301	008NA060001	H91E0	F227006	365
3	FR8212006	73NA050004	H9410	F227007	200
4	FR8202002	74NA060006 P1	H9130	F227008	478
5	FR8202002	74NA060006 P2	H9130	F227009	1025
6	FR8201741	038NA050003	H9140	F227010	29
7	FR8201670	007NA060003 P1	H9120	F22711	765
8	FR8201670	007NA060003 P2	H9120	F22711	765
9	FR2600988	058NA050003	H91D0	F22711	40
10	FR8201749	038NA050001	H91F0	F22711	86
11	FR8202002	74NA060006 P2	H9430	F22711	1316
12	FR8201749	007NA050001 P1	H91F0	F22711	105
13	FR8201749	007NA050001 P2	H91F0	F22711	262
14	FR8201686	026NA060002	H9110	F22711	176
15	FR8201670	007NA060008	H9120	F22712	5
16	FR2100273	008NA040002	H9190	F22712	7
17	FR4310027	025NA060005 P1	H9130	F22712	8
18	FR4310027	025NA060005 P2	H9130	F22712	67
19	FR8201741	038NA050003	H9180	F22712	13

Table 13. Continued.

N	Site code	Forest contract/parcel	Habitat type code	Action code	Action cost €/ha/y
20	FR8201741	038NA050003	H9410	F22712	6
21	FR8201764	042NA060001	H9190	F22712	2
22	FR4202002	068NA050001	H9130	F22712	148
23	FR4202002	068NA050003	H9130/9110	F22712	154
24	FR4202002	068NA050004	H9130	F22712	118
25	FR4202002	068NA050006	H91EO/9130	F22712	136
26	FR4202002	068NA050007	H9130	F22712	63
27	FR4202002	068NA050008	H9130	F22712	139
28	FR4202002	068NA050009	H9110/9130	F22712	105
29	FR4202002	068NA050010	H9110/9180	F22712	136
30	FR4202002	068NA050011	H9130	F22712	160
31	FR4202002	068NA050002	H9410/ 9110	F22712	122
32	FR4202002	068NA050005	H9410	F22712	66
33	FR4301342	070NA060001 P1	H91F0	F22712	67
34	FR4301342	070NA060001 P2	H91F0	F22712	61
35	FR4301348	090NA060002	H9110/9130	F22712	67
36	FR8201688	026NA050005	H9130	F22712	12

6.1.2 Telephone survey and field inventory (pre/post-intervention)

Telephone survey

The pre-intervention of biodiversity indicators have been often inventoried within Natura 2000 contracts at intervention or contract level. The existence and accessibility of such data has been queried via telephone survey. Such data has been collected for the majority of contract. However, for remaining intervention parcels, the value of relevant biodiversity indicators has been simulated based on the type and intensity of conducted actions. For instance, pre-intervention percentage of invasive species and deadwood has been simulated using the type (eliminating invasive species) and intensity (basal area of eliminated invasive trees) of the conducted action.

Field inventory using Point-centred quarter method (PCQM)

The sampling targets the status of biodiversity indicators directly affected by conducted actions. These indicators, in turn, allow second evaluation of the conservation status (degree of conservation) at intervention level. While plot-based methods are widely used in forest inventories, plot-less methods or distance-based approaches often appeared to be more efficient and cost-effective (Cantarello et al., 2008). Plot-less methods involve measuring distances for a random sample of trees, typically along a transect, and recording the characteristics of interest for this sample such as dead or alive, large... The advantage to using plot-less methods rather than standard plot-based techniques is that they tend to be more efficient. Plot-less methods are faster, require less equipment, and may require fewer workers. Another advantage is that results are not dependent on the size of quadrants. However, the main advantage is speed.

The point-centred quarter method (PCQM) is one of such plot-less method. Motivation to use of the PCQM rather than other plot-less methods or rather than plot-based methods not only includes the statistically sound quantitative basis, but also logistic factors such as ease and speed with which the fieldwork can be executed in the Natura 2000 sites (Cantarello and Newton, 2008). The question, then, is whether accuracy is sacrificed in the process. Many authors tested and stated that PCQM is highly recommended and used in forest inventory (Cottam & Curtis 1956, Beasom and Haucke, 1975, Engeman et al. 1994). For instance, Cantarello and Newton (2008) applied and evaluated plot-based and PCQ Method for evaluating biodiversity indicators in Natura 2000 sites of Italy and UK. The authors used management units having more than 1 hectare as a sampling units and then 30 of them were randomly selected. For plot-based method, a 0.25-ha (50m*50m) north-orientated plot was randomly established within each sampling units. For plot-less method, a 100-m north-orientated transect with three fixed sample points, 50 m distant from each other, was randomly established within each sampling units. The area around each sample point was divided into four equiangular sectors. In each sector, the distance to the four closest trees was measured. As such, a total number of 16 trees per sample point and 48 trees per transect was sampled. The choice of method had no significant impact on the results obtained for the six indicators common to both methods on both sites while the plot-less method appeared to be the most time efficient. However, both approaches appeared to be different in terms of the extent of variation within the indicators, the number of indicators that can be assessed and the spatial extent of surveyed forest. Even though, for a limited number of indicators and limited areas of intervention parcels in this research, the plot-less method sounds fully relevant.

The PCQM of Cottam and Curtis (1956) is one of the plot-less methods that has been considered very efficient in characterizing vegetation, while minimizing damage to the forest understory and to estimate standing dead tree (Cantarello and Newton, 2008; Rheinhardt et al. 1997).

Field inventory using Line intersect method (LIM)

Since its original description by Warren and Olsen (1964), the line intersect method (LIM) has been extensively used for measuring the quantity of wood lying on the ground (fallen deadwood). The original application was for the estimation of logging residues, further developed by Bailey (1970). Van Wagner (1968) and Brown (1971) described the method’s use for measuring forest fuels. De Vries (1973) investigated its mathematical basis in depth, and Affleck et al (2005) carried out a series of simulation studies. More recently, it has been used to monitor productivity, stand structure, and wildlife. Many authors recommended this method to have a quick and efficient assessment of the down woody materials (DWM) (Affleck 2008, Bobiec 2011).

Line-intersect sampling (LIS) is now commonly used to estimate the attributes of dead, down woody debris in forest ecosystems. In LIS, a transect, is established over an area based on a predetermined design. All dead, down woody debris crossed by the line transect that meets the sampling criteria is assessed; the specific attributes required to meet the study objectives are then recorded for each piece (e.g., species, piece tilt, diameter at transect, and length). Here, we describe some basic design principles for sampling randomly distributed DWD:

(1) sampling precision depends mainly on the total length of line (i.e., the length of a sampling unit x the number of sampling units; the length and arrangement of individual lines is immaterial); (2) the size of the area to be sampled is theoretically irrelevant and (3) sampling precision increases as the number of DWD intersections per unit length of sample line increases (i.e., precision depends on the density and size of pieces). These principles do not take into account sampling costs and do not necessarily apply when DWD has a clustered, directional, or otherwise non-random spatial pattern (Storm direction).

Sampling unit and sampling design

Contact with the selected sites managers has been crucial to obtain all necessary information and maps as well as to coordinate field work and finally to find the right intervention parcel or sampling unit where the actions have been carried out. All these steps have been done during telephone surveys and field work.

Natura 2000 sites ⇨ Contracts ⇨ Intervention parcel or sampling unit

Following PCQM method, a transect covering the whole area is randomly chosen in each parcel. In order to better cover sample unit, a diagonal orientation was employed or even several transects were used when the form of the unit was irregular. On each sampling unit, a 100 or 200 -m orientated transect with ten or twenty sample points, 10 or 20 m distant from each other, was established within each Sample Unit. The area around each sample point was divided into four equiangular sectors. In each equiangular sector or quadrant all relevant biodiversity indicators has been measured: (1) distance from the plot centre to the nearest tree and its diameter; (2) distance from the plot centre to the nearest standing dead tree and its diameter and (3) distance from the plot centre to the very large living tree and its diameter. Finally, according to line intersect method, the diameter and the dead cause of fallen dead wood crossed by the transect was measured. The species have been annotated in all the cases. In each transect, it has been taken 10 or 20 sample point, depending on the size of the area since the transect cannot exceed the totality of the parcel length, assuring the inventory was carried out within the evaluated area with a sufficient number of sample point (often assumed to be no less than 10).

Selection of biodiversity indicators

The forest structure or the management of a forest for silvicultural or biodiversity purposes requires structural parameters such as density and basal area. In the same way, assessing the conservation status of the habitat requires biodiversity related indicators such as standing and fallen deadwood and percentage of invasive, non typical and typical species. In our study, we decided to measure only a view parameters related to conservation purposes and the Carnino protocol. The Diameter at Breast Height (DBH) of each tree (dead or alive) >7.5 cm dbh was measured using calipers. Large trees was defined as those with dbh >50 cm. However, for fallen dead wood, the diameter and the dead cause on the fallen dead wood crossed by the transect was measured.

6.2 Data processing

6.2.1 Calculating biodiversity indicators using PCQM and LIM data

PCQM data

In order to evaluate the degree of conservation in an intervention level, relevant biodiversity indicators should be calculated based on existing or sampled data. As such, a new score will be calculated for each criterion and finally, the total score of conservation will be estimated.

Pollard (1971) derived an unbiased estimate of the absolute population density using PCQM. It has the advantage that it can be used to determine confidence intervals for the density estimate. It based on the angle-order method. The area around each sample point was divided into four equiangular sectors (90°) and the distance to the four closest trees of interest (alive, large and snags) in each sector was measured. As a practical matter, a major reason to use the PCQM is its efficiency, which is at odds with substantial sampling effort. Additionally, sample points along the transect should be sufficiently far apart so that the same tree is not sampled at two adjacent transect points. Dahdouh-Guebas and Koedam (2006) suggest that it may be preferable to establish a consistent distance limit for the sampling point to the nearest individual rather than to consider the same individual twice. As such, a distance limit is established for reasons of efficiency [often called truncated sampling]. In practice vacant quarters, i.e., quadrants containing no tree may occur. In such cases, the calculation of the absolute density must be corrected, since a density calculated from only those quarters containing observations will overestimate the true density. Patil et al. (1979) derived formula for the density and its variance. Using this formula with the modifications in Patil et al. (1982) leads to the following. Let w be the upper limit for the radius beyond which one does not search. Let n be the number of sample points and let n1 denote the number of sample points with observations, i.e., points where the distance to the nearest organism does not exceed w. So there are $n_0 = n - n_1$ sample points without observations. The data are the order statistics R(k) , where $k = 1; : : : ; n_1$. Tree density and basal area percentage (alive, large and standing dead tree estimation per hectare) was calculated using the formula as reported in Mitchell (2007) as follows:

Then

$$\hat{\lambda} = \frac{n_1}{n}\left(\frac{n_1^{2/3}-1}{n_1 \pi R^2_{([n_1^{2/3}])}}\right). \tag{10}$$

An estimate of the variance is given by

$$\mathrm{Var}(\hat{\lambda}_t) = \frac{\hat{\lambda}_t^2}{n_1^{2/3}} + \hat{\lambda}_t^2\left(\frac{1}{n_1}-\frac{1}{n}\right)\left(1+\frac{1}{n_1^{2/3}}\right). \tag{11}$$

For large samples, the endpoints of a $(1-\alpha)100\%$ confidence interval for the density λ are well-approximated by

$$C_1 = \hat{\lambda} + z_{\frac{\alpha}{2}}\sqrt{\mathrm{Var}(\hat{\lambda}_t)} \quad \text{and} \quad C_2 = \hat{\lambda} + z_{1-\frac{\alpha}{2}}\sqrt{\mathrm{Var}(\hat{\lambda}_t)}. \tag{12}$$

with R = the point-to-tree distance at point i in quarter j in meter.

LIM data

The line intersect method is best pictured as a strip sample of infinitesimal width. The data collected are the diameters of the dead wood pieces > 7.5 cm at their points of intersection with the transect, the same as the PCQM. The transect is really a vertical plane, and the tally in effect collects a series of circular cross-sectional areas from the intersected wood pieces. Of course the actual cross-sectional areas are really ellipses of various shapes (except when the intersection is exactly at right angles), but, for convenience, a factor derived from probability theory allows the areas to be summed as circles. The sum of cross-sectional areas is then divided by the length of the transect; at this point the result is in terms of cross-sectional area per unit length of the transect. Multiplying both numerator and denominator by width converts the line sample into a strip sample (Van Wagner and Wilson, 1976), and the result can then be quoted as volume per unit of ground area. The basic equation (Van Wagner, 1968) is

$$V = (\pi^2/8L)\ \Sigma\ d^2 \tag{13}$$

where V is volume per unit area,
d is piece diameter at intersection,
L is length of sample line.

Based on these calculation methods, a script running on R© was developed by Damien Marage (AgroParisTech) and Kevin Mitchell (Department of Mathematics and Computer Science, Hobart and William Smith Colleges) for this purpose. It provides a report file and figures for all biodiversity indicators: alive tree density and basal area, standing dead tree density and basal area and large tree density (Table 14). Similarly, the existing and telephone survey data were used to estimate relevant biodiversity indicators before intervention at intervention level (Table 15).

Table 14. The post-treatment values of studied biodiversity indicators at intervention level based on sampled and existing data

N	Forest contract/parcel	Habitat type code	Action code	% Non typical tree species	% invasive tree/shrub species	Number of Large Tree (per ha)	Number of standing Dead Tree (per ha)	Number of fallen dead Wood (per ha)	Number of Dead Wood (per ha)	% typical species
1	073NA050003 P1	9130	F227005	0.0	0.0	0.6	8.4	3.0	11.4	100.0
2	008NA060001	91E0	F227006	0.0	0.0	2.0	4.3	1.0	5.3	100.0
3	73NA050004	9410	F227007	11.5	0.0	0.0	0.0	1.0	1.0	88.5
4	74NA060006 P1	9130	F227008	100.0	0.0	10.0	12.0	8.0	20.0	0.0
5	74NA060006 P2	9130	F227009	97.5	0.0	0.0	0.0	8.0	8.0	2.5
6	038NA050003	9140	F227010	0.0	0.0	4.8	21.3	5.8	27.1	100.0
7	007NA060003 P1	9120	F22711	0.0	0.0	0.0	0.0	0.0	5.0	100.0
8	007NA060003 P2	9120	F22711	0.0	0.0	0.0	0.0	0.0	5.0	100.0
9	058NA050003	91D0	F22711	5.6	0.0	0.0	9.0	1.0	10.0	94.4
10	038NA050001	91F0	F22711	5.9	69.9	28.0	0.0	6.0	6.0	94.1
11	74NA060006 P2	9430	F22711	37.5	0.0	0.0	1.0	8.0	9.0	62.5
12	007NA050001 P1	91F0	F22711	8.1	72.5	12.0	6.0	3.0	9.0	91.9
13	007NA050001 P2	91F0	F22711	0.7	12.2	0.0	0.0	6.0	6.0	99.3
14	026NA060002	9110	F22711	53.5	0.0	0.0	7.0	4.0	11.0	46.5
15	007NA060008	9120	F22712	0.0	0.0	31.0	4.0	0.0	4.0	100.0
16	008NA040002	9190	F22712	0.9	0.0	2.8	34.0	0.0	34.0	99.1
17	025NA060005 P1	9130	F22712	19.1	0.0	52.0	2.0	0.0	7.0	80.9
18	025NA060005 P2	9130	F22712	14.0	0.0	52.2	0.0	4.0	4.0	86.0
19	038NA050003	9180	F22712	16.9	0.0	0.0	7.0	4.0	11.0	83.1
20	038NA050003	9410	F22712	0.0	0.0	3.8	27.2	0.9	28.1	100.0
21	042NA060001	9190	F22712	3.9	0.0	5.0	0.0	0.0	0.0	96.1
22	068NA050001	9130	F22712	0.0	0.0	73.9	0.0	7.0	7.0	100.0
23	068NA050003	9130/9110	F22712	4.5	0.0	75.0	8.0	0.0	8.0	95.5
24	068NA050004	9130	F22712	9.6	0.0	48.1	0.0	4.0	4.0	90.4
25	068NA050006	91EO/9130	F22712	23.2	0.0	44.0	14.0	9.6	23.6	76.8
26	068NA050007	9130	F22712	0.0	0.0	39.1	6.0	4.0	10.0	100.0
27	068NA050008	9130	F22712	5.3	0.0	50.8	6.6	8.2	14.8	94.7
28	068NA050009	9110/9130	F22712	0.0	0.0	39.5	7.0	0.0	7.0	100.0
29	068NA050010	9110/9180	F22712	9.3	0.0	58.7	3.0	8.0	11.0	90.7
30	068NA050011	9130	F22712	0.0	0.0	92.5	5.4	0.8	6.2	100.0
31	068NA050002	9410/ 9110	F22712	12.5	0.0	35.7	3.3	1.4	4.7	87.5
32	068NA050005	9410	F22712	11.4	0.0	25.7	2.8	2.5	5.3	88.6
33	070NA060001 P1	91F0	F22712	0.0	0.0	12.5	0.0	0.0	0.0	100.0
34	070NA060001 P2	91F0	F22712	0.0	0.0	12.1	0.0	0.0	0.0	100.0
35	090NA060002	9110/9130	F22712	11.6	0.0	46.5	21.0	8.0	29.0	88.4
36	026NA050005	9130	F22712	6.7	0.0	8.8	0.0	0.0	0.0	93.3

Table 15. The pre-treatment values of studied biodiversity indicators at intervention level based on existing data

N	Forest contract/parcel	Habitat type code[3]	Action code	% Non typical tree species	% invasive tree/shrub species	Number of Large Tree (per ha)	Number of standing Dead Tree (per ha)	Number of fallen dead Wood (per ha)	Number of Dead Wood (per ha)	% typical species
1	073NA050003 P1	9130	F227005	0.0	0.0	0.6	0.0	0.0	0.0	100.0
2	008NA060001	91E0	F227006	0.0	0.0	2.0	4.3	0.0	4.3	100.0
3	73NA050004	9410	F227007	11.5	0.0	0.0	0.0	0.0	0.0	88.5
4	74NA060006 P1	9130	F227008	100.0	0.0	10.0	0.0	0.0	0.0	0.0
5	74NA060006 P2	9130	F227009	100.0	0.0	0.0	0.0	0.0	0.0	0.0
6	038NA050003	9140	F227010	0.0	0.0	4.8	5.0	0.8	5.8	100.0
7	007NA060003 P1	9120	F22711	5.5	0.0	0.0	0.0	0.0	0.0	94.6
8	007NA060003 P2	9120	F22711	5.0	0.0	0.0	0.0	0.0	0.0	95.0
9	058NA050003	91D0	F22711	4.0	0.0	0.0	9.0	1.0	10.0	96.0
10	038NA050001	91F0	F22711	5.2	79.2	32.7	0.0	4.0	4.0	94.9
11	74NA060006 P2	9430	F22711	86.7	0.0	0.0	0.0	0.0	0.0	13.3
12	007NA050001 P1	91F0	F22711	8.1	64.7	15.5	3.0	0.0	3.0	91.9
13	007NA050001 P2	91F0	F22711	0.7	60.8	22.7	0.0	5.0	5.0	99.3
14	026NA060002	9110	F22711	53.5	2.7	0.0	7.0	0.0	7.0	46.5
15	007NA060008	9120	F22712	0.0	0.0	31.0	4.0	0.0	4.0	100.0
16	008NA040002	9190	F22712	0.9	0.0	2.8	34.0	4.0	38.0	99.1
17	025NA060005 P1	9130	F22712	19.1	0.0	52.0	2.0	7.0	9.0	80.9
18	025NA060005 P2	9130	F22712	14.0	0.0	0.0	6.0	4.0	10.0	86.0
19	038NA050003	9180	F22712	16.9	0.0	0.0	7.0	4.0	11.0	83.1
20	038NA050003	9410	F22712	0.0	0.0	3.8	27.2	0.9	28.1	100.0
21	042NA060001	9190	F22712	3.9	0.0	5.0	0.0	0.0	0.0	96.1
22	068NA050001	9130	F22712	0.0	0.0	73.9	0.0	7.0	7.0	100.0
23	068NA050003	9130/9110	F22712	4.5	0.0	75.0	8.0	0.0	8.0	95.5
24	068NA050004	9130	F22712	9.6	0.0	48.1	0.0	4.0	4.0	90.4
25	068NA050006	91EO/9130	F22712	23.2	0.0	44.0	14.0	9.6	23.6	76.8
26	068NA050007	9130	F22712	0.0	0.0	39.1	6.0	4.0	10.0	100.0
27	068NA050008	9130	F22712	5.3	0.0	50.8	6.6	8.2	14.8	94.7
28	068NA050009	9110/9130	F22712	0.0	0.0	39.5	7.0	0.0	7.0	100.0
29	068NA050010	9110/9180	F22712	9.3	0.0	58.7	3.0	8.0	11.0	90.7
30	068NA050011	9130	F22712	0.0	0.0	92.5	5.4	0.8	6.2	100.0
31	068NA050002	9410/ 9110	F22712	12.5	0.0	35.7	3.3	1.4	4.7	87.5
32	068NA050005	9410	F22712	11.4	0.0	25.7	2.8	2.5	5.3	88.6
33	070NA060001 P1	91F0	F22712	0.0	0.0	12.5	0.0	0.0	0.0	100.0
34	070NA060001 P2	91F0	F22712	0.0	0.0	12.1	0.0	0.0	0.0	100.0
35	090NA060002	9110/9130	F22712	11.6	0.0	46.5	21.0	8.0	29.0	88.4
36	026NA050005	9130	F22712	6.7	0.0	8.8	0.0	0.0	0.0	93.3

[3] According to interpretation manual of European Union habitats - EUR 27 DG Environment, Nature and biodiversity, 2007, 144p.

6.2.2 Calculating degree of conservation before and after intervention

Based on calculated biodiversity indicators, Carnino scoring system was employed to calculate the total score or the degree of conservation before and after intervention for each inventoried parcel (Table 16).

Table 16. Degree of conservation before and after intervention with an average time span of 7 years at intervention level

N	Site code	Contract number	Intervention Parcel	Degree of conservation Initial score	Degree of conservation Final score	Conservation status
1	FR8212006	073NA050003	P	60	80	Favorable
2	FR2100301	008NA060001	P	70	90	Favorable
3	FR8212006	073NA050004	P	55	65	Unfavorable
4	FR8202002	074NA060006	P1	40	60	Unfavourable
5	FR8202002	074NA060006	P2	20	40	Unfavourable
6	FR8201741	038NA050003	P	86	88	Favorable
7	FR8201670	007NA060003	P1	55	78	Favorable
8	FR8201670	007NA060003	P2	55	78	Favorable
9	FR2600988	058NA050003	P	75	75	Favorable
10	FR8201749	038NA050001	P	63	65	Unfavourable
11	FR8202002	074NA060006	P2	30	50	Unfavourable
12	FR8201749	007NA050001	P1	63	65	Unfavourable
13	FR8201749	007NA050001	P2	68	75	Favorable
14	FR8201686	026NA060002	P	48	50	Unfavourable
15	FR8201670	007NA060008	P	98	98	Favorable
16	FR2100273	008NA040002	P	90	90	Favorable
17	FR4310027	025NA060005	P1	85	85	Favorable
18	FR4310027	025NA060005	P2	93	93	Favorable
19	FR8201741	038NA050003	P1	83	83	Favorable
20	FR8201741	038NA050003	P2	88	88	Favorable
21	FR8201764	042NA060001	P	78	78	Favorable
22	FR4202002	068NA050001	P	95	95	Favorable
23	FR4202002	068NA050003	P	98	98	Favorable
24	FR4202002	068NA050004	P	93	93	Favorable
25	FR4202002	068NA050006	P	85	85	Favorable
26	FR4202002	068NA050007	P	100	100	Favorable
27	FR4202002	068NA050008	P2	95	95	Favorable
28	FR4202002	068NA050009	P	100	100	Favorable
29	FR4202002	068NA050010	P	95	95	Favorable
30	FR4202002	068NA050011	P	100	100	Favorable
31	FR4202002	068NA050002	P	93	93	Favorable
32	FR4202002	068NA050005	P	93	93	Favorable
33	FR4301342	070NA060001	P1	80	80	Favorable
34	FR4301342	070NA060001	P2	80	80	Favorable
35	FR4301348	090NA060002	P	95	95	Favorable
36	FR8201688	026NA050005	P	75	75	Favorable

Finally, all data obtained using existing database, telephone survey and field inventory was incorporated into a data set. The data set developed in the last section were then used to obtain a large number of pre-treatment, treatment (dose) and response variables at intervention level. A correlation analysis was then employed to retain all variables having significant effect on the response variable. The analysis was also used to find the most correlated forms of variables. The results are presented in Table 17.

Table 17. Significant pre-treatment and treatment (dose) variables affecting the degree of conservation of studied intervention parcel

N	Site code	Contract number	Action Parcel	Cost €/ha/y	Action type F227005	Action type F227011	Action type F227012	Percent change of conservation status	Alsace Region
1	FR8212006	073NA050003	P	400	1	0	0	0.33	0
2	FR2100301	008NA060001	P	365	1	0	0	0.29	0
3	FR8212006	073NA050004	P	200	1	0	0	0.18	0
4	FR8202002	074NA060006	P1	478	1	0	0	0.50	0
5	FR8202002	074NA060006	P2	1025	1	0	0	1.00	0
6	FR8201741	038NA050003	P	29	1	0	0	0.02	0
7	FR8201670	007NA060003	P1	765	0	1	0	0.42	0
8	FR8201670	007NA060003	P2	765	0	1	0	0.42	0
9	FR2600988	058NA050003	P	40	0	1	0	0.00	0
10	FR8201749	038NA050001	P	86	0	1	0	0.03	0
11	FR8202002	074NA060006	P2	1316	0	1	0	0.67	0
12	FR8201749	007NA050001	P1	105	0	1	0	0.03	0
13	FR8201749	007NA050001	P2	262	0	1	0	0.10	0
14	FR8201686	026NA060002	P	176	0	1	0	0.05	0
15	FR8201670	007NA060008	P	5	0	0	1	0.00	0
16	FR2100273	008NA040002	P	7	0	0	1	0.00	0
17	FR4310027	025NA060005	P1	8	0	0	1	0.00	0
18	FR4310027	025NA060005	P2	67	0	0	1	0.00	0
19	FR8201741	038NA050003	P1	13	0	0	1	0.00	0
20	FR8201741	038NA050003	P2	6	0	0	1	0.00	0
21	FR8201764	042NA060001	P	2	0	0	1	0.00	0
22	FR4202002	068NA050001	P	148	0	0	1	0.00	1
23	FR4202002	068NA050003	P	154	0	0	1	0.00	1
24	FR4202002	068NA050004	P	118	0	0	1	0.00	1
25	FR4202002	068NA050006	P	136	0	0	1	0.00	1
26	FR4202002	068NA050007	P	63	0	0	1	0.00	1
27	FR4202002	068NA050008	P2	139	0	0	1	0.00	1
28	FR4202002	068NA050009	P	105	0	0	1	0.00	1
29	FR4202002	068NA050010	P	136	0	0	1	0.00	1
30	FR4202002	068NA050011	P	160	0	0	1	0.00	1
31	FR4202002	068NA050002	P	122	0	0	1	0.00	1
32	FR4202002	068NA050005	P	66	0	0	1	0.00	1
33	FR4301342	070NA060001	P1	67	0	0	1	0.00	0
34	FR4301342	070NA060001	P2	61	0	0	1	0.00	0
35	FR4301348	090NA060002	P	67	0	0	1	0.00	0
36	FR8201688	026NA050005	P	12	0	0	1	0.00	0

7. Estimating provision cost function of biodiversity protection using inverse ordinal dose-response model

As stated earlier, provision cost function relates the cost of provision to quantity of output or impact of conducted actions and relevant ecological variables like initial degree of conservation. As explained earlier, provision cost function is obtained by inversing impact assessment model (ordinal logit dose-response model). However, it is easier to estimate provision cost function since its dependent variable is continuous:

$$C_{LR} = f(Y_f\text{-}Y_i,\ T;\ Z) \qquad (14)$$

As such, provision cost function also allows assessing the impact of conducted action. In this equation, the change made in the degree of conservation (Y_f-Y_i) is a better impact indicator than its post-treatment value (Y_f) since the difference mitigates pre-treatment differences between intervention parcels. As such, more change costs logically more. In addition, provision cost depends on the type of conducted action (T) and ecological pre-treatment conditions (Z). Concerning action type, a distinction should be made between maintenance (action F22712) and restoration actions (F22705 and F22711). Restoration actions can improve the degree of conservation while maintenance actions are conducted just to maintain the current degree of conservation.

The cost of restoration actions depend on the change that is to be made in the degree of conservation. In addition, for the same action and the same change in the degree of conservation, the less conservation degree is, the more the restoration costs. Indeed, initial the degree of conservation represents a good proxy for ecological pre-treatment conditions (Z). As such, the most efficient specification of these three variables, (Y_f-Y_i), T and Z for restoration actions (F22705 and F22711) obtained as:

$$\left(\frac{Y_f - Y_i}{Y_i}\right) * \text{Restoration action type}$$

By contrast, the cost of maintenance actions seems to be invariant with the change in the degree of conservation or even with initial degree of conservation but depends rather on the physical amount of actions e.g., Number of Large Trees (NLT) maintained for biodiversity. Thus, the most efficient specification for maintenance action (action F22712) obtained as:

$$NLT * \text{Maintenance action type}$$

However, effect of all significant pre-treatment conditions such as region effect has been also integrated in the model. As such, different specifications of the model have been estimated.

The estimation was done using Nonlinear GMM procedure in SAS software. The best specification appeared to be as following:

$$C_{LR} = a.NLT.F22712 + b.\left(\frac{Y_f - Y_i}{Y_i}\right).F22705 + c.\left(\frac{Y_f - Y_i}{Y_i}\right).F22711 + d.Alsace \qquad (15)$$

With

C_{LR}: Long run provision cost

NLT: Number of large trees

F22712: Binary variable representing the action F22712 “enhancing senescent (large) trees”

Y_f : Final degree (score) of conservation (after intervention)

Yi : Initial degree (score) of conservation (before intervention)

F22705: Binary variable representing the action F22705 “Marking, felling or pruning without production objective”

F22711: Binary variable representing the action F22711 “Eliminating or limitation of invasive species”

Alsace: Binary variable representing Alsace as a region in French

a, b, c and d: Model parameters

The result of estimation, statistics of model and fit diagnostics were presented in Tables 18, 19 and 20, respectively.

Table 18. Nonlinear GMM estimation result of provision cost function (Parameter Estimates)

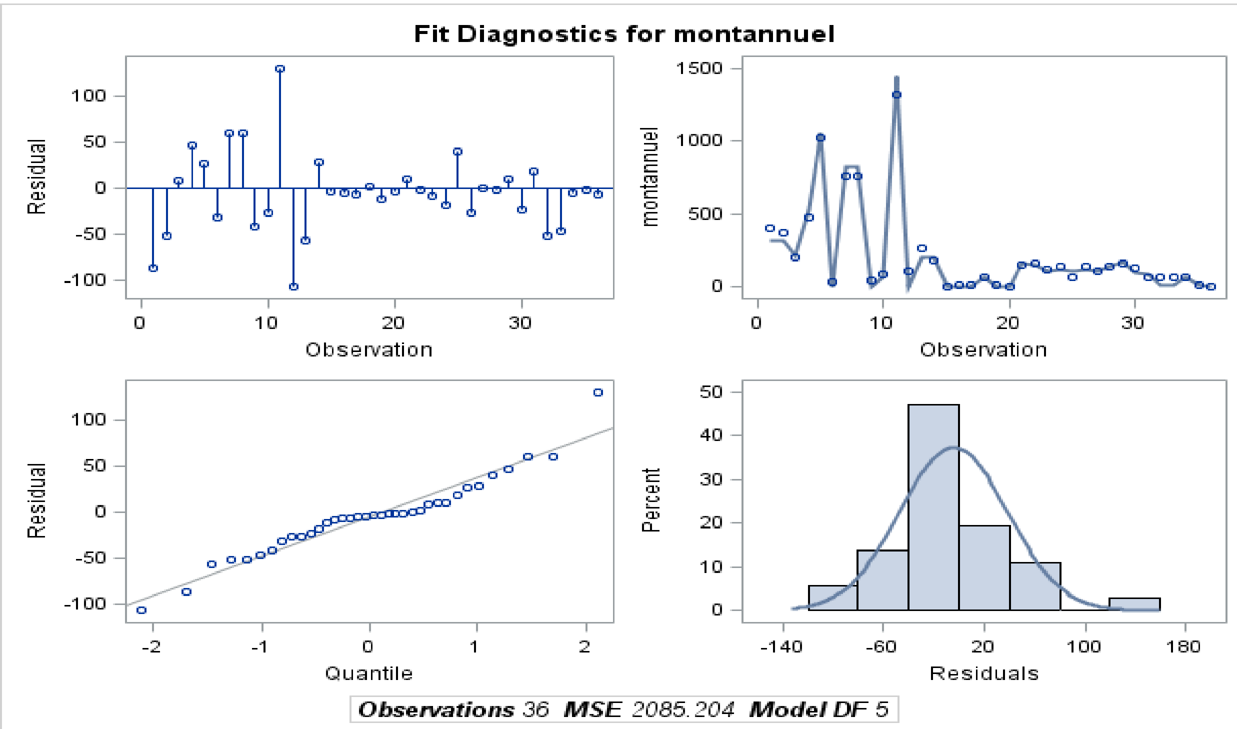
Parameter	Estimate	Approx Std Err	t Value	Approx Pr > |t|
Cte	-1.61515	11.3158	-0.14	0.8874
F227012*NLT	1.369403	0.2328	5.88	<.0001 *
F227005* $\frac{Y_f - Y_i}{Y_i}$	1053.404	99.0691	10.63	<.0001 *
F227011* $\frac{Y_f - Y_i}{Y_i}$	2067.811	218.1	9.48	<.0001 *
Alsace	52.24383	9.2805	5.63	<.0001 *

*__All variables are significant (except the constant) at the 1% level.__

Table 19. Nonlinear GMM Summary of statistics of the model

Equation	Model DF	DF	SSE	MSE	Root MSE	R-Square	Adj R-Sq
Provision cost function	**5**	**31**	**64641.3**	**2085.2**	**45.6640**	**0.9798**	**0.9771**

Table 20. Fit diagnostics (goodness of fit) of the model



As shown above, all variables and model performance statistics are significant at the 1% level. Adjusted coefficient of determination, Adj R-Sq was estimated 97.71 percent. As such, the model explains 97.71 percent of the data variance. "RMSE" (Root Mean Square Error) or "SEE" (the Standard Error of the Estimate) was estimated 45.66 €. Fit diagnostics also show clearly the goodness of fit. Residuals have no heteroscedasticity and represent a normal distribution with a zero average and constant variance. In general, the model estimation seems satisfactory.

The proposed model distinguishes between maintenance (action F22712) and restoration costs (F22705 and F22711). Restoration cost can improve the degree of conservation while maintenance action is often employed to maintain the degree of conservation. Except the constant, all variables appeared to be significant at the 1% level with expected signs. Restoration actions (F22705 and F22711) conducted within French Natura 2000 network were significantly effective in promoting degree of conservation of biodiversity at 1% level. The efficiency of these actions in promoting degree of conservation was estimated to be 1%. As such, 1% raise in biodiversity costs will result in 1% increase in conservation status. However, for the same change percent of the degree of conservation, action F22705 costs almost half compared to action F22711. Finally, the average provision cost to enhance by 1% the conservation score of biodiversity through both actions was estimated at 15.6 € ha^{-1} y^{-1}. Similarly, maintenance action (F22712) conducted within French Natura 2000 network were significantly effective in maintaining the degree of conservation of biodiversity at 1% level. The average provision cost to maintain the conservation score of biodiversity through action F22712 was estimated to be 1.4 € per tree per year. The model incorporate also a binary variable called “Alsace” being 1 for all intervention parcels at Alsace region and equals zero otherwise. The estimated parameter is around 52 € since Alsace has had the highest payment ceiling for action F22712 during the study period.

Figure 3 shows the relation between restoration cost and the change made in degree of degree of conservation.

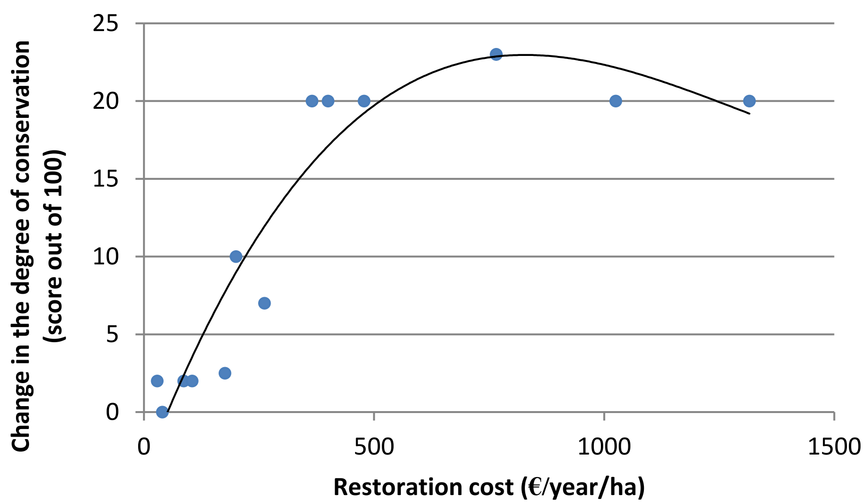


Figure 3. Relation between restoration cost and the change made in degree of degree of conservation

As shown in Figure 3, more change costs logically more. The saturation point cannot be analyzed precisely in the figure since the type of actions and pre-treatment conditions are not controlled for. However, saturation effect is quite expected as the change was made only to restoration cost while other variables were kept constant.

Restoration cost is by nature operational cost and thus, depends on the type of action conducted. As such, for the same change in the degree of conservation, action F22705 costs in average half compared to action F22711 (compare parameters b and c in the model). In addition, for the same action and the same change in the degree of conservation, the less the initial degree of conservation is the more restoration costs (Figure 4).

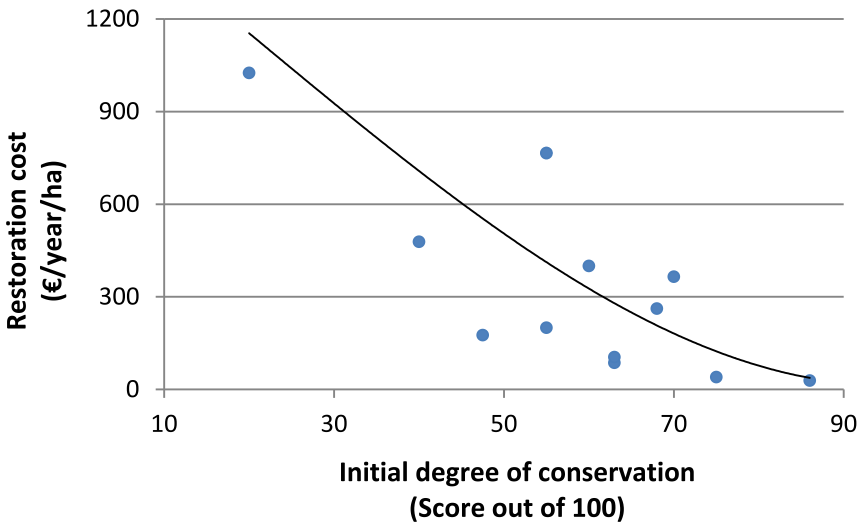


Figure 4. The relation between restoration cost and initial degree of conservation

As such, it may be more economical to launch conservation programs in due time i.e., before the degree of conservation decreases more or exceeds its economic threshold (restoration cost is more than restoration benefits). In addition, opportunity cost is often higher where the degree of conservation is low since in this region extensive forestry should compete with intensive agriculture or forestry. For instance, 19 percent of intervention parcels are in unfavourable status, indicating a late and more costly intervention compared to favorable status.

In other words, the more an ecosystem is degraded the more the restoration will cost. Let’s define the degree of degradation as opposed to the degree of conservation (100-degree of conservation). Figure 5 present the relation between restoration cost and degree of degradation.

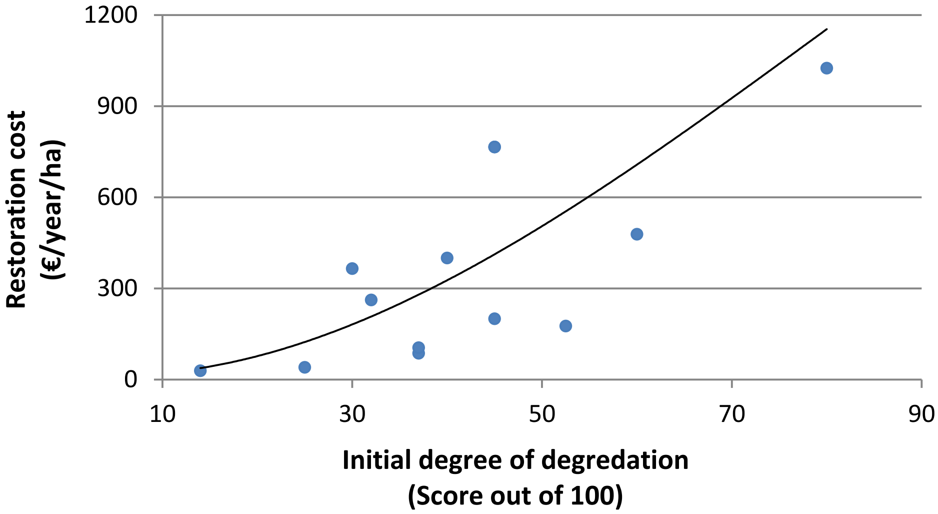


Figure 5. The relation between restoration cost and initial degree of degradation

We find once more the same concept in Figure 5; for the same action and the same change in the degree of conservation, the more an ecosystem is degraded the more the restoration will cost. In addition, initial degree of conservation represents a good proxy for pre-treatment conditions. As such, we regressed the restoration cost versus the change percent of the degree of conservation i.e., the change of degree of conservation divided by initial degree of conservation (Figure 6). This relation present much more linear correlation compared to other relations studied above.

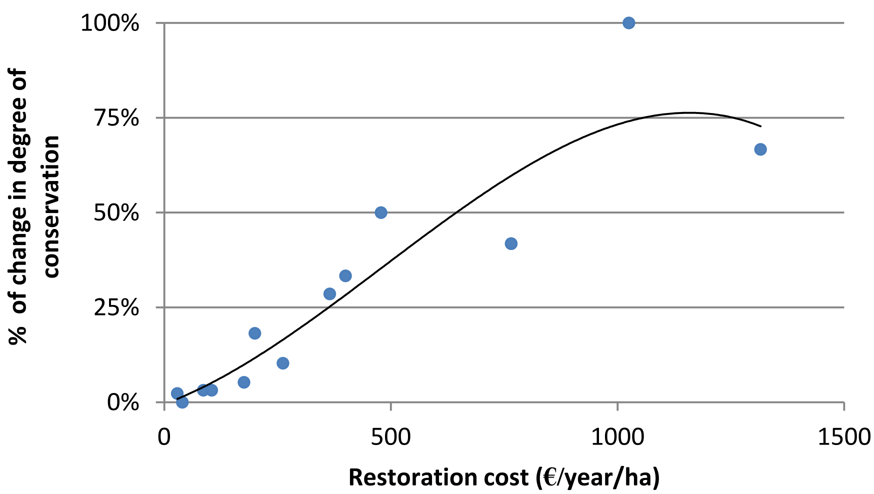


Figure 6. The relation between restoration cost and change percent in the degree of conservation

By contrast, maintenance cost is paid just to maintain the current degree of conservation. Maintenance cost is by nature opportunity cost depending on forgone use of certain resources for the sake of biodiversity. As such, it's often invariant against change percent of the degree of conservation. For action F22712, maintenance cost is proportional to number of large trees maintained individually or in group " (îlot") as habitat trees (Figure 7).

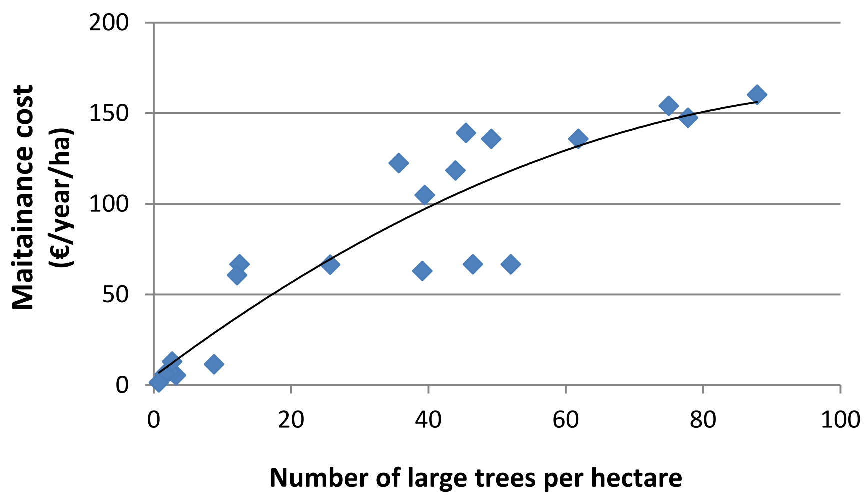


Figure 7. The relation between maintenance cost and number of large trees per hectare

As shown in Figure 7, there is an almost linear relation between maintenance costs and the number of eligible large trees maintained for biodiversity. However, regardless of the number of biodiversity trees, there is certain payment ceiling per hectare and per period varying according to national or regional regulations. For instance, in France this payment ceiling was around 4000 €/ha for a period of 30 years which is now the same sum at national level. However, this payment ceiling was previously different by regions and Alsace region had the higher payment (4000 €/ha/30y) ceiling compared to other French regions (2000 €/ha/30y). This is why this binary variable was introduced in the model.

Conclusion and discussion

An inverse dose-response model based only on participating Natura 2000 site was selected and applied in this research to assess both provision cost and biodiversity action impact. Such method was also justified and applied by Bia and Mattei (2007). They showed that regression based methods (non-experimental design) such as MRA can be also applied only for participating sites or enterprises (Natura 2000). As such, they applied MRA for estimating a dose-response model relating response variable (outcome or impact indicator) to pre-treatment and treatment (dose) variables. Thus, they inferred the intervention impact using the coefficient of the treatment variable in the model. Different from Bia and Mattei (2007) model, in this research the endogeneity problem was also taken into account using GMM estimator. However, comparable studies seemed to be rare. The synthesis work of Kleijn and Sutherland (2003) gives a best overview on impact assessment methods used for biodiversity actions. Kleijn and Sutherland (2003) reviewed 62 studies evaluating the impact of European agri-environment scheme on biodiversity in five EU countries and Switzerland. The majority of these studies had not adequate research design to assess reliably the effectiveness of the schemes. “The commonest experimental design (37% of the studies) was a comparison of biodiversity in agri-environment schemes and control areas”. However, there is a risk of selection bias due to pre-intervention differences between agri-environment schemes and control sites. In such cases the schemes area are likely to have a higher biodiversity at the outset compared to the control. The authors concluded that the lack of robust evaluation studies does not allow a general judgment of the effectiveness of European agri-environment schemes. They proposed that ecological evaluations must become an integral part of any scheme, including the collection of baseline data, the random placement of scheme and control sites in areas with similar initial conditions, and sufficient replication. As long as, such facility are not available, dose-response model with GMM estimator seems to be the most suitable method.

Results revealed that biodiversity actions conducted within French Natura 2000 network were significantly effective in promoting degree of conservation of biodiversity at 1% level. However, it was not possible to conclude statistically on the effectiveness of action F22712 in maintaining degree of conservation. Indeed, the correlation of this action with number of eligible large trees was logically much higher that its correlation with initial or final degree of conservation. However, maintaining large trees contribute obviously to maintain the degree of conservation. Similarly, Kleijn and Sutherland (2003) in their review of 62 relevant studies concluded that agri-environment schemes were generally effective in promoting the richness or abundance (impact indicator) of species as 54% of the examined species groups were increased compared to control groups. However, 23% of the examined species showed no change at all in response to agri-environment schemes.

The efficiency of restoration actions in promoting degree of conservation of biodiversity was estimated to be 1%. As such, 1% raise in biodiversity costs will result in 1% increase in degree of conservation.

For the same change percent of the degree of conservation, action F22705 costs almost half compared to action F22711. The average provision cost to enhance 1% the conservation score of biodiversity through the both actions was estimated 15.6 € ha^{-1} y^{-1}. Similarly, the average provision cost to maintain the conservation score of biodiversity through action F22712 was estimated to be 1.4 € ha^{-1} y^{-1} per tree. However, these costs may not be considered only as provision cost of biodiversity protection since the latter contribute to promoting other forest goods and services e.g. tourism and recreation, water quality, flood control and so on. In addition, biodiversity investments in Natura 2000 sites can be particularly important for local and regional economic development (See RDP objectives) since they provide direct and indirect local benefits such as employment. As such, the ecological and socio-economic benefits of biodiversity protection can be larger than the associated costs (Gantioler et al., 2010).

The results must be interpreted as a short term assessment of biodiversity actions indicating first impacts of biodiversity actions. Although the number of observation per action type is relatively small, this analysis provides certain results which should be tested on larger set of observations.

The methodology proposed in this research seems applied and promising. It presents several implications, of which:

- Provision cost function may be helpfully used to estimate compensation payments for any increase in biodiversity conservation score;

- Payments based on result (given change in biodiversity score) help to reduce information asymmetry between the both sides of contract especially when monitoring is rather complicated;

- Cost-effectiveness has not yet figured highly in European biodiversity conservation research and policy- making (Watzold and Schwerdtner, 2004). Such impact assessment and provision costs models may contribute to deal with cost-effectiveness issues in relation to conservation policies;

- Such impact assessment and provision costs models may contribute to deal with the cost-effective allocation of conservation fund in space and time;

- Applying such impact assessment models permit to identify the most effective and efficient market based instruments (e.g., compensation scheme, taxes, fees and charges, subsidies/support, tradable permits and eco-labelling and financial mechanisms) in different contexts;

- The proposed model distinguishes between maintenance (action F22712) and restoration costs (F22705 and F22711);

This methodology can be usefully applied for

- other European countries
- other biodiversity actions
- other forest externalities e.g., water regulation under Water Framework Directive
- or other environmental services under Rural Development Regulation;

Provision cost function provide a common basis (e.g., given change in biodiversity score) for estimating comparable marginal cost and value;

Comparable estimation of provision cost and value for forest externalities may provide helpful information both for forest managers and policy makers.

Obtained results should be interpreted as a short term evaluation with small number of observations. Long term monitoring of biodiversity actions and their impacts with larger number of observation at intervention level will enable a more complete and precise evaluation of intervention impacts. The existing and sampled data have not necessarily obtained through similar sampling method with comparable precision. As such, the estimated change percent of biodiversity score may not be quite exact but it seems to be the most precise measure accessible at the finest scale. However, a permanent monitoring of conservation status at micro level over time permits more precise comprehension and assessment of the impact of biodiversity actions. Different market based instruments (MBIs) can be applied for biodiversity conservation. These results was obtained using compensation payments under Natura 2000 contract. However, other MBIs such as taxes, fees and charges, subsidies/support, tradable permits and eco-labelling and financial mechanisms (e.g. green venture capital funds) can be used. The choice of MBIs may also affect provision cost as well as effectiveness and efficiency of biodiversity actions. Generalizing this pilot research to include other biodiversity actions and other EU countries may provide relevant information for policy and decision makers.

Appendix 1: Questionnaire of telephone survey in French

Questionnaire d'enquête sur les contrats Natura 2000 forestiers auprès des animateurs des sites

Objectif :

Mesurer l’effet de certaines actions en faveur de la biodiversité (Code PDRH, F22705, F22711 et F22712) sur les indicateurs correspondants.

Evaluer la perception des animateurs des sites quant à la contractualisation sur les sites Natura 2000

Coordonnées du contrat en question :

N° contrat

N° du site Natura 2000

Nom de l'animateur

Coordonnées de l'animateur

Commune majoritaire sur laquelle le contrat est passé

I) Questionnement général :

Méthode d’évaluation initiale de l’état de conservation (avant élaboration de contrat)

1. D'après FSD 2. À dire d'expert 3. Méthode Carnino 4. Autres (à préciser)

Période de mise en œuvre des actions du contrat ?

Numéro d’identification de la parcelle où l’action est mise en œuvre

(cadastrale ou le n° de la parcelle forestière)

Comment obtenir le plan du parcellaire forestier ?

Quelles sont les coordonnées de la personne correspondante ?

II) Mesurer l'effet de certaines actions en faveur de la biodiversité sur les indicateurs correspondants :

Est-ce que les indicateurs ci-après sont mesurés avant et/ou après la mise en œuvre des actions dans les parcelles affectées par le contrat ?

Si oui, quelles sont les valeurs associées et l'an de la mesure ?

Nombre à l'hectare des très gros bois (ayant le diamètre 65< pour les feuillus et 70< pour les résineux)

Nombre à l'hectare des arbres morts (ayant le diamètre 35<=)

Nom et pourcentage de la surface terrière de l'espèce exotique envahissante

Si non, est-il possible de mesurer ces indicateurs dans les parcelles où les actions ont été menées ? (Au mois de Mai) Dans ce cas-là, qui est-ce la personne correspondante (morale ou physique)? Quelles sont ses coordonnées ?

Si l'action F22705 (travaux de marquage, d'abattage ou de taille sans enjeu de production) est mise en œuvre dans les parcelles affectées par le contrat ; s'applique-t-elle principalement aux petits bois ou gros bois ?

III) Evaluer la perception des animateurs des sites quant à la contractualisation sur les sites Natura 2000

Est-ce que les actions en faveur de la biodiversité affectent visiblement d'autres services écosystemiques (par exemple, quantité et qualité de l'eau etc.) ? comment (positivement ou négativement) ? exemple le plus probant ?

Est-ce que ce contrat a été renouvelé ? Si oui, quand ? si non, pourquoi ?

D'après vous, quelles sont les motivations principales pour la contractualisation ?

Les montants par barème ou hors barème des actions sont-ils suffisants pour mener les actions ?

A part les coûts directs payés sous le contrat pour la mise en œuvre des actions, quel sont les coûts ignorés ? (par exemple, les coûts indirects ou d'opportunité etc.)

A part le paiement pour la mise en oeuvre des actions, quels sont les autres bénéfices de ces actions? (par exemple, avoir plus facilement un label de gestion durable comme FSC etc.)

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