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DE LORRAINE



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THÈSE
pour l'obtention du grade de
Docteur en Sciences Économiques

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UNIVERSITÉ DE LORRAINE

**Incentive payments for biodiversity conservation: A dynamic
and spatial analysis**

par

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Résumé de la thèse

Contexte et motivation

La biodiversité, définie comme la somme de toutes les “plantes, animaux, champignons et microorganismes sur Terre, leur variation génotypique et phénotypique, ainsi que les communautés et les écosystèmes dont ils font partie” (Dirzo et Raven, 2003) est actuellement érodée à un rythme alarmant, et ce, de façon globale. Les taux actuels d’extinction observés dépassent les projections établies sur la base d’études paléontologiques (Barnoski *et al.*, 2011).

Les changements environnementaux anthropiques tels que la perte et la dégradation des habitats liés aux changements d’usage du sol ont été identifiés comme étant les principaux moteurs de cette érosion de biodiversité, au même titre que la surexploitation des ressources, la pollution, l’introduction d’espèces envahissantes et le changement climatique (Foley *et al.*, 2005 ; Pereira *et al.*, 2012 ; Allan *et al.*, 2014 ; Newbold *et al.*, 2015).

Or, la biodiversité - *via* sa contribution au fonctionnement des écosystèmes - soutient la fourniture de nombreux services écosystémiques (SE) bénéficiant aux sociétés humaines. La biodiversité est ainsi essentielle à la fourniture de nombreux SE, tels que les services d’approvisionnement (e.g., aliments, fibres, carburants, etc.), de régulation (e.g., maintien de la qualité de l’eau, séquestration du carbone atmosphérique), de soutien (e.g., cycles biogéochimiques) et les services culturels (e.g., valeurs esthétiques et récréatives) (MEA, 2005 ; Cardinale *et al.*, 2012 ; Hooper *et al.*, 2012 ; Harrison *et al.*, 2014). Outre l’immense valeur sociale de ces SE dérivant indirectement de la biodiversité, la biodiversité présente également une valeur de non-usage liée notamment aux valeurs d’option et d’existence (Jobstvogt *et al.*, 2014 ; Bartkowski, 2017). Ainsi, l’érosion de la biodiversité est susceptible d’impacter fortement le bien-être des sociétés humaines

(Allan *et al.*, 2015).

Cette érosion de la biodiversité d'origine anthropique peut alors sembler contradictoire. Ce phénomène est néanmoins expliqué par la théorie économique. En effet, plusieurs biens et services découlant de la biodiversité – telle que la séquestration du carbone atmosphérique – illustrent le concept de bien public, tandis que d'autres – tels que les stocks de gibier ou les ressources halieutiques – illustrent la notion de bien commun. Dans le premier cas, la fourniture du bien est soumise au problème de *passager clandestin* (Olson, 1935) et la théorie économique prédit la fourniture de celui-ci en quantité sous-optimale. Dans le second cas, la théorie économique prédit l'épuisement du bien, non-exclusif mais rival, comme l'explique Hardin (1968) dans la *Tragédie des communs*. Dans les deux cas, l'appauvrissement des ressources et la conservation insuffisante de la biodiversité correspondent à des équilibres de Nash. La protection de la biodiversité correspond, quant à elle, à une externalité positive résultant de comportements "vertueux" de la part des agents qui, selon la théorie économique, ne devraient pas émerger.

Cela montre la nécessité pour le régulateur public de prendre en charge la conservation de la biodiversité. Trois types d'instruments peuvent être mis en place afin d'internaliser cette externalité et atteindre l'optimum social (Weitzman, 1974) : (1) les instruments réglementaires, consistant à imposer une norme uniforme à tous les agents, (2) les instruments basés sur le marché, fournissant une incitation économique aux agents à modifier leur comportement et (3) les instruments de type "gérance environnementale", consistant à fournir des informations aux agents afin d'influencer leurs décisions et les conduire à adopter des comportements vertueux.

La conservation de la biodiversité a traditionnellement reposé sur la mise en place d'aires naturelles protégées ainsi que d'instruments réglementaires. A l'heure actuelle, les décideurs publics recourent encore largement à ce type d'instruments (MEA, 2005) : ainsi, 18% de la surface terrestre de l'Union européenne (UE) est protégée dans le cadre de la politique Natura 2000 (N2K) et le réseau terrestre N2K français est constitué de 1 766 sites N2K couvrant 7 048 086 hectares (INPN, 2017). Des instruments réglementaires tels que le régime de protection stricte défini par la Directive Habitats en Union eu-

ropéenne ou l'Endangered Species Act aux USA sont actuellement en vigueur. Cependant, la mise en place de ces instruments de type "command and control" peut être combinée à celle d'instruments de conservation incitatifs. Les mesures agri-environnementales (MAE) mises en oeuvre dans les États Membres de l'UE dans le cadre de la politique agricole commune (PAC), les aides accordées dans le cadre du Conservation Reserve Program aux USA ainsi que le programme PSA mis en oeuvre en forêt au Costa Rica sont autant d'exemples d'instruments incitatifs existants visant, entre autres, à la conservation de la biodiversité. Ces dispositifs illustrent plus précisément le concept de paiements pour services environnementaux (PSE).

Ces instruments de type PSE ont suscité un fort intérêt de la part des décideurs publics au cours des dernières décennies. Des dispositifs de conservation illustrant ce concept de PSE ont été mis en place dans un grand nombre de pays industrialisés et en développement dans les paysages agricoles et forestiers. Des MAE ont été mises en oeuvre dans les 28 États membres de l'UE et le budget de la Commission européenne destiné aux MAE s'élevait à près de 20 milliards d'euros pour la période 2007-2013. L'utilisation des PSE est également encouragée dans le cadre de la stratégie biodiversité de l'UE à l'horizon 2020. Malgré cet enthousiasme de la part des décideurs publics et l'utilisation croissante des mécanismes incitatifs pour la conservation de la biodiversité, on sait peu de choses sur la coût-efficacité des programmes mis en oeuvre (Ansell *et al.*, 2016), en particulier dans les forêts (mais voir Barton *et al.*, 2009). De nombreuses études ont étudié l'efficacité écologique des MAE dans les paysages agricoles (e.g., Kleijn *et al.*, 2004 ; Kleijn *et al.*, 2006 et Besnard et Secondi, 2014). Cependant, très peu de ces études prennent en considération les coûts de conservation et moins de 15% d'entre elles incluent une évaluation de la coût-efficacité du dispositif (Ansell *et al.*, 2016). Or, la mise en oeuvre de paiements incitatifs pour la conservation de la biodiversité est coûteuse et la promotion de ce type d'instruments par les décideurs publics devrait reposer sur une connaissance solide de leur coût-efficacité.

Problématique et objectifs de la thèse

Dans ce contexte, l'objectif de cette thèse est d'étudier la définition de paiements incitatifs pour la conservation de la biodiversité d'un point de vue empirique et théorique. Dans ce travail, nous visons également à intégrer de façon pertinente les processus

écologiques spatiaux et dynamiques inhérents à la biodiversité terrestre dans les modèles économiques que nous développons.

Dans le premier chapitre de cette thèse, nous évaluons la coût-efficacité des paiements pour contrats N2K mis en place en forêt en France. Le caractère insuffisant de la définition de ces paiements et leur mauvaise calibration montre la nécessité de repenser la définition de ces dispositifs d'incitations.

Dans le deuxième et troisième chapitre de cette thèse, nous étudions la définition de paiements incitatifs efficaces et coût-efficaces de façon théorique et conceptuelle, tout en prenant en compte les principaux enjeux posés par la définition de paiements incitatifs pour la conservation de la biodiversité terrestre.

Le chapitre 2 étudie la possibilité de définir des paiements incitatifs différenciés à destinations des propriétaires privés lorsque les coûts et bénéfices de conservation sont hétérogènes et inobservables pour l'agence de conservation. Ce chapitre s'intéresse donc particulièrement à l'impact de l'asymétrie d'information et du phénomène d'anti-sélection sur la définition des paiements.

Dans le chapitre 3, nous nous intéressons principalement à l'impact du changement climatique sur la définition de paiements incitatifs coût-efficaces. Dans ce chapitre, nous développons un modèle écologique-économique intégré, dynamique et spatialement explicite, nous permettant d'étudier la coût-efficacité relative de différents types de design, impliquant différents degrés de ciblage et de différenciation des paiements de conservation.

Chapitre 1 : Évaluation de la coût-efficacité d'une politique de conservation de la biodiversité - Une analyse bio-économétrique des contrats Natura 2000 en forêt

Dans ce premier chapitre, nous étudions la coût-efficacité des paiements incitatifs pour la conservation de la biodiversité mis en place en forêt en France, à savoir les contrats N2K forestiers. De telles approches volontaires pour la conservation de la biodiversité en forêt ont été mises en oeuvre dans d'autres États Membres de l'UE tels que le Danemark, l'Allemagne ou la Slovaquie dans le cadre de N2K (Anthon *et al.*, 2010; Ecochard *et al.*, 2017). Plusieurs études ont examiné empiriquement les déterminants

de la participation des propriétaires forestiers et de leur consentement à recevoir (e.g., Boon *et al.*, 2010; Prokofieva *et al.*, 2013). Cependant, à notre connaissance, très peu d'études évaluent quantitativement et empiriquement la mise en oeuvre de la politique N2K et des instruments de conservation incitatifs associés en forêt.

Dans ce travail, nous exploitons la base de données OSIRIS¹ portant sur les instruments de conservation incitatifs mis en place en forêt dans le cadre de N2K en France entre 2007 et 2010, à savoir les chartes et contrats N2K forestiers, et évaluons empiriquement et quantitativement la coût-efficacité des contrats. Nous adoptons une approche *ex ante* : nous utilisons les données relatives aux espèces et habitats ciblés par chaque contrat et quantifions le niveau "d'output biodiversité" fourni par celui-ci par le biais d'un indice de biodiversité.

Nous estimons simultanément une fonction de coût de conservation de la biodiversité et l'ensemble de production de biodiversité et de bois. Grâce aux données portant sur les caractéristiques du contrat, nous contrôlons l'effet des mesures de conservation mises en place ainsi que l'effet du type de bénéficiaire du contrat (e.g., personne de droit public ou privé) dans notre estimation. Nous prenons également en compte l'impact de l'hétérogénéité écologique et socio-économique sur le coût du contrat et la fourniture de biodiversité.

Tout d'abord, nos résultats soulignent la qualité des plans de gestion (i.e., documents d'objectifs) mis en place à l'échelle du site N2K : ces derniers assurent la mise en oeuvre cohérente des différents types de mesures de conservation. En outre, ces plans de gestion fournissent un cadre utile pour la protection et la conservation des espèces et des habitats vulnérables.

Nos résultats permettent également de formuler des recommandations concernant le ciblage des contrats futurs. Nous montrons par exemple l'intérêt de favoriser la mise en oeuvre de contrats par des bénéficiaires publics dans un contexte de forte pression foncière. En effet, nous montrons que ces derniers sont en mesure de supporter des coûts d'opportunité plus élevés que les propriétaires privés en termes de valeur foncière. Ce-

¹Cette base de données nationale est gérée par l'agence de services et de paiement et contient l'ensemble des informations relatives à la gestion des aides au développement rural (caractéristiques du bénéficiaire, localisation, type et durée de l'engagement, montant de l'aide accordée, etc.). Nous disposons d'une extraction anonymisée de cette base portant uniquement sur les chartes et contrats N2K forestiers signés entre 2007 et 2010.

pendant, les propriétaires forestiers privés présentent des caractéristiques intéressantes en termes de coût-efficacité par rapport aux bénéficiaires publics. Cette coût-efficacité des propriétaires privés n'est cependant pas mise à profit dans le cadre actuel du dispositif de contrats N2K forestiers. On observe en effet un faible niveau de participation, en particulier de la part des propriétaires forestiers privés.

Cette faible participation pourrait s'expliquer par la mauvaise calibration des paiements à l'heure actuelle. En effet, nous montrons que les coûts d'opportunité liés aux pertes de revenus sylvicoles ne sont pas pris en compte dans la définition du paiement, et ce, malgré la substituabilité entre production de bois et conservation de la biodiversité dans le cadre des contrats N2K forestiers en France. Une telle définition est susceptible d'entraîner l'établissement de contrats présentant un faible niveau d'additionnalité, en favorisant la participation d'agents caractérisés par de faibles coûts d'opportunité et en écartant ceux ayant des coûts d'opportunité élevés liés aux revenus sylvicoles.

Enfin, nous montrons que la conservation de la biodiversité dans le cadre des contrats N2K forestiers en France est caractérisée par des rendements d'échelle croissants. Cela suggère l'établissement de contrats plus ambitieux permettant la conservation de plusieurs habitats et espèces au sein d'un même contrat, sous réserve d'exigences écologiques compatibles.

Chapitre 2 : Définir des contrats de conservation espèces-spécifiques dans un paysage hétérogène avec des coûts et bénéfices de conservation inobservables

La définition de contrats pour la conservation de la biodiversité nécessite, dans certains cas, de prendre en considération une asymétrie d'information portant sur les coûts et les bénéfices de la conservation. Une agence de conservation pourrait, par exemple, souhaiter cibler les paiements vers des propriétés présentant une richesse spécifique élevée ou hébergeant une espèce rare. Cependant la mise en place d'activités de monitoring *ex ante* afin d'observer la richesse spécifique de la propriété ou la présence d'une telle espèce cible *in situ* pourrait engendrer des coûts de transaction élevés pour l'agence de conservation. Dans ce contexte, nous étudions la définition de paiements de conservation efficaces dans le cadre d'un programme de conservation espèce-spécifique. Nous examinons la possibilité de définir des contrats de conservation différenciés tout en tenant

compte du caractère inobservable et hétérogène des coûts et bénéfices de conservation.

Dans ce chapitre, nous considérons un modèle principal-agent, dans lequel un régulateur (i.e., une agence de conservation) délègue la mise en place de mesures de conservation à un propriétaire privé dans le cadre d'un programme espèce-spécifique.

Le régulateur a deux objectifs interdépendants : la mise en place d'un habitat favorable à l'espèce (découlant directement de la surface mise en conservation) et, la protection effective des individus de l'espèce (*via* la mise en oeuvre de mesures de conservation dans les propriétés hébergeant des individus). Le coût de conservation inobservable du propriétaire privé est bidimensionnel et correspond à la somme des coûts d'opportunité et des coûts de protection. L'absence/présence d'individus de l'espèce cible dans une propriété est signalée par le niveau de coût de protection du propriétaire. Nous traduisons ce problème de conservation par un modèle principal-agent à *valeur commune*, dans lequel la préférence du régulateur pour les deux types de bénéfices de conservation est explicitement prise en compte : l'utilité du principal augmente en fonction de la surface mise en conservation et du niveau de coût de protection. Nous comparons la performance des contrats définis sur la base d'un modèle à valeur commune et d'un modèle de sélection adverse classique, en termes de différenciation des paiements et de coût-efficacité, par le biais de simulations numériques à l'échelle du paysage.

Nous contribuons à la littérature appliquant la théorie des contrats à la définition de PSE en tenant compte du caractère non observable des bénéfices de conservation ainsi que de la multidimensionnalité de l'asymétrie d'information, ce qui a été rarement fait dans cette littérature (voir Feng, 2007). En outre, nous développons un cadre analytique, plus complet et mieux adapté à la définition de contrats optimaux dans le cadre de programmes de conservation espèce-spécifique.

Nous démontrons analytiquement la possibilité de définir des paiements de conservation différenciés en dépit d'une situation de "non-responsiveness", menant généralement à des équilibres mélangeants. Les résultats des simulations numériques à l'échelle du paysage montrent que les contrats définis sur la base d'un modèle à valeur commune peuvent présenter une meilleure performance que ceux dérivés d'un modèle de sélection adverse classique. En effet, d'une part le modèle à valeur commune permet, dans un contexte d'asymétrie d'information, la définition de contrats plus proches des contrats optimaux

définis dans un contexte d'information parfaite. D'autre part, les contrats dérivés d'un modèle à valeur commune peuvent présenter des performances intéressantes en termes de coût-efficacité. Cependant, il n'est pas toujours coût-efficace de définir des contrats sur la base d'un modèle à valeur commune.

La considération implicite d'aspects spatiaux dans notre travail nous permet de formuler des recommandations sur le type d'instruments à mettre en œuvre selon le contexte de conservation (i.e., probabilité de présence de l'espèce cible, additionnalité de la protection de l'espèce, orientation et potentiel du paysage en termes de production sylvicole ou agricole, etc.). Nous montrons, par exemple, qu'il est particulièrement efficace de fonder la définition des contrats de conservation sur un modèle à valeur commune en cas d'additionnalité intermédiaire de la protection de l'espèce. Le degré de rareté de l'espèce (indiqué par sa probabilité de présence) favorise également la coût-efficacité des contrats définis sur la base d'un modèle à valeur commune. Cependant, il serait fortement inefficace de fonder la définition des contrats de conservation sur un modèle à valeur commune dans le cas d'un faible niveau d'additionnalité. Enfin, un programme de conservation espèce-spécifique ne devrait pas reposer sur des contrats de conservation incitatifs dans le cas d'un niveau élevé d'additionnalité de la protection de l'espèce.

Chapitre 3 : Coût-efficacité des dispositifs de paiements pour la conservation de la biodiversité dans un contexte de changement climatique

La définition d'instruments de conservation incitatifs coût-efficaces à l'échelle du paysage dans un contexte dynamique a été étudiée, notamment par la littérature en écologie-économie. Cependant, à notre connaissance, cette littérature ne prend pas en compte l'impact de la dynamique induite par le changement climatique à l'échelle du paysage sur la définition des paiements. Or, le changement climatique a été identifié comme l'une des principales menaces pour la conservation de la biodiversité (Sala *et al.*, 2000). De plus, malgré le fort potentiel souligné par la littérature des instruments de types PSE pour la mise en place de réseaux de conservation dans un contexte de changements globaux (Donald et Evans, 2006, Heller et Zavaleta, 2009), il semble nécessaire d'adapter les dispositifs existants afin de répondre spécifiquement aux enjeux posés par le changement climatique (Donald et Evans, 2006).

Dans le troisième chapitre de cette thèse, nous nous appuyons sur les travaux en modélisation écologique-économique intégrée portant sur la coût-efficacité des paiements de conservation et étudions la définition de paiements de conservation dans un contexte de changement climatique.

Nous développons un modèle écologique-économique intégré, dynamique et spatialement explicite à l'échelle du paysage. Nous étudions les performances relatives en termes de coût-efficacité de quatre types de paiements : i) des paiements spatialement uniformes basés sur les actions et attribués aux propriétaires mettant en place des mesures de conservation, ii) des paiements spatialement différenciés dont le montant dépend du caractère favorable du climat caractérisant les différentes parties du paysage considéré, iii) des paiements ciblés attribués aux propriétaires mettant en place des mesures de conservation uniquement si l'espèce est présente sur leur propriété ou dans une propriété adjacente, iv) des paiements combinant différenciation et ciblage. Nous cherchons à identifier l'impact de variables économiques et écologiques clés sur la coût-efficacité des différents types de paiements, telles que le degré d'hétérogénéité des coûts de conservation dans le paysage, la capacité de dispersion de l'espèce, la stabilité du climat, etc. La modélisation de paiements ciblés nécessite la prise en compte de boucles de rétroaction entre les modèles économiques et écologiques que nous intégrons dans notre travail.

Nos résultats montrent que la mise en oeuvre de paiements ciblés et/ou différenciés permet la réalisation de gains significatifs en termes de coût-efficacité par rapport aux paiements uniformes. En outre, nous montrons que des arbitrages de type connectivité/surface, mis en évidence dans le cadre de paysages dynamiques, sont aussi observés dans un contexte de changement climatique. Enfin, nous montrons que les avantages apportés par les paiements ciblés en termes de connectivité du réseau de conservation sont amoindris par une diminution de la stabilité du climat dans le paysage.

La nature conceptuelle de ce travail limite la pertinence directe de nos résultats en termes de politique de conservation. Cependant, nous apportons une première réflexion quant à la définition de paiements pour la conservation de la biodiversité dans un contexte de changement climatique. De plus, le modèle écologique que nous utilisons est un modèle général pouvant être appliqué aux espèces présentant une dynamique de métapopulation. Il serait donc intéressant d'appliquer ce travail à des données réelles. Cela nous permettrait de formuler des recommandations plus fines en termes de politiques de conservation

pour les études de cas considérées.

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Introductory Chapter

1 Context and motivation

1.1 Biodiversity erosion and its implications for human societies: the necessity to encourage biodiversity conservation

Biodiversity, defined as the sum of all “plants, animals, fungi, and microorganisms on Earth, their genotypic and phenotypic variation, and the communities and ecosystems of which they are a part” (Dirzo and Raven, 2003), is being globally eroded at an alarming pace. In the 20th century birds, mammals and amphibian species have become extinct at a rate of 48 extinctions per million species years (Pereira *et al.*, 2012) and current observed species extinction rates generally exceed what would be expected according to fossil records (Barnoski *et al.*, 2011). Anthropogenic environmental changes such as habitat change and degradation linked to land-use decisions have been identified as the major drivers of biodiversity loss, along with human-induced overexploitation of resources, pollution, introduction of invasive species and climate change (Foley *et al.*, 2005; Pereira *et al.*, 2012; Allan *et al.*, 2014; Newbold *et al.*, 2015).

Yet, biodiversity underpins — *via* its contribution to ecosystem functions — the supply of numerous ecosystem services² (ES) benefiting to human societies such as provisioning (e.g. food, fibers, fuel, and genetic resources), regulating (e.g., water purification and regulation, carbon sequestration), supporting (e.g., nutrient cycling) and cultural services (e.g., aesthetic values and recreation) (MEA, 2005; Cardinale *et al.*, 2012; Hooper *et al.*, 2012; Harrison *et al.*, 2014). Apart from the tremendous social value of these ES indirectly deriving from biodiversity, biodiversity is also granted non-use values such as option³ and existence⁴ values (Jobstvogt *et al.*, 2014; Bartkowski, 2017). Biodiversity erosion is therefore susceptible to significantly impact well-being of human societies in the future through the loss of ES (Allan *et al.*, 2015).

Thus, the depletion of biodiversity by humankind might seem paradoxical. This phenomenon is nevertheless explained by economic theory. Indeed, several goods and

²Understood as benefits humans obtain from ecosystems.

³The option value of an environmental good corresponds to the economic value (i.e., willingness to pay) attributed by human societies to the possibility of making use of this good in the future. There is no consensus on whether this option value belongs to the use value or non-use value of the good (Wattage *et al.*, 2008).

⁴The existence value is part of the non-use value of an environmental good and can be defined as the economic value (i.e., willingness to pay) of this good apart from any use by human societies.

services provided by biodiversity illustrate the concept of public goods, such as carbon sequestration, while others, such as game and fisheries, illustrate the concept of common goods. In the former case, the provision of the good – because of its non-excludable character nature – is subject to the *free-rider problem* (Olson, 1935) and economic theory predicts the sub-optimal provision of the good. In the latter case, economic theory predicts the depletion of the – non-excludable but rival – good as described by Hardin (1968) in the *Tragedy of the commons*. In both cases, the sub-optimal provision and depletion of biodiversity correspond to Nash equilibria, where no player has an incentive to deviate from her strategy. On the contrary, the protection of biodiversity corresponds to a positive externality resulting from “virtuous” behaviors which, according to economic theory, should not be expected from rational agents.

This shows the need for the public regulator to take responsibility for the conservation of biodiversity. Three types of instruments can be used by the regulator to internalize this externality and reach the social optimum (Weitzman, 1974): (1) regulatory instruments which consist in imposing a uniform standard on all agents, (2) market-based instruments providing economic incentives to agents to change their behavior and (3) information and awareness instruments consisting in the provision of information to agents in order to influence their decisions and lead them to adopt virtuous behaviors.

1.2 From “command-and-control” approaches to incentive payments for biodiversity conservation

The conservation of biodiversity has traditionally been based on “command-and-control” instruments such as natural protected areas and regulatory instruments. The creation of national parks, such as Yellowstone in 1872 in the United States of America (USA), allowed the preservation of exceptional biodiversity, emblematic and charismatic species as well as the conservation of traditional landscapes.

Biodiversity conservation today still heavily relies on natural protected areas (MEA, 2005) and regulatory tools. For instance, 18% of the European Union’s (EU) terrestrial surface is protected under the Natura 2000 (N2K) policy and the French terrestrial network is made of 1,766 N2K sites covering 7,048,086 hectares (INPN, 2017). Regulatory instruments such as the strict protection regime defined by the EU Habitats Directive or

the Endangered Species Act in the USA are currently in force. However, “command-and-control” instruments can be combined with incentive conservation instruments, allowing the conservation of ordinary and exceptional biodiversity on private land. For example, N2K sites are not strict natural reserves. On the contrary they are managed in a sustainable development perspective requiring both economic as well as ecological aspects to be accounted for (preamble and Art 2., Habitat Directive) and incentive conservation instruments can be implemented within N2K sites. Agri-environment schemes (AES) implemented in the EU member states within the framework of the common agricultural policy (CAP), aids granted under the Conservation Reserve Program in the United States, as well as the PSA program implemented in forests in Costa Rica are examples of existing incentive instruments aiming, among others, at biodiversity conservation. These devices more specifically illustrate the concept of payments for environmental services (PES).

These PES-type incentives have received increasing attention from research as well as from public decision-makers in recent decades. As mentioned above, devices illustrating this concept of PES have been implemented for biodiversity conservation in a large number of developed and developing countries in both agricultural and forest landscapes. AES have been implemented in the 28 member states of the EU and the EU Commission spent nearly €20 billion for the 2007 – 2013 period on such schemes. The use of this type of instrument is also encouraged under the EU 2020 Biodiversity Strategy. Despite this enthusiasm on the part of decision-makers and the increasing use of incentive-based mechanisms for biodiversity conservation over the past decade, little is known about the cost-effectiveness⁵ of implemented schemes (Ansell *et al.*, 2016), especially in forests (but see Barton *et al.*, 2009). Numerous studies have investigated the ecological effectiveness of AES implemented in agricultural landscapes (e.g., Kleijn *et al.*, 2004; Kleijn *et al.*, 2006 and Besnard and Secondi, 2014) but very few of them account for AES costs and less than 15% include any measure of cost-effectiveness (Ansell *et al.*, 2016). Yet, implementing incentive payments for biodiversity conservation is costly and public support for these instruments should be based on a sound knowledge of their cost-effectiveness.

⁵Understood as the maximization of conservation outcome for a given cost or the minimization of cost for a given conservation outcome.

2 Research questions and objectives of the thesis

In this context, **the objective of this thesis is to study the definition of incentive payments for biodiversity conservation from an empirical and theoretical point of view.** In this work, we also aim to account, in a relevant way, for **spatial and dynamic ecological processes inherent to terrestrial biodiversity**⁶ in the economic models that we develop.

In the first chapter of this thesis, we empirically assess the cost effectiveness of incentive payments for biodiversity conservation implemented in French forests, namely N2K contracts. Such contracts for forest biodiversity conservation have been implemented in other EU member states like Denmark, Germany or Slovakia (Anthon *et al.*, 2010; Ecochard *et al.*, 2017). Several studies have empirically investigated the determinants of forest owners' participation in voluntary conservation approaches as well as their willingness to accept (e.g. Boon *et al.*, 2010; Prokofieva *et al.*, 2013). However, to our knowledge very few studies quantitatively and empirically assessed the implementation of the N2K policy and incentive-based conservation instruments in forests (see Chapter 1 for details). Our results underline the inadequacy of the current definition of payments for N2K contracts and their poor calibration.

Chapter 1

Assessing the cost-effectiveness of a biodiversity conservation policy: A bio-econometric analysis of Natura 2000 contracts in forest

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While conservation schemes are usually qualitatively and globally assessed, we empirically and quantitatively assess the cost-effectiveness of N2K forest contracts by undertaking an *ex ante* approach. We use data related to targeted species and habitats at the individual contract level to quantify the biodiversity output of a contract through a biodiversity index.

⁶Hereafter referred to as “biodiversity”.

We simultaneously estimate a cost function for biodiversity conservation and the production set of biodiversity output and timber, while controlling for conservation measures as well as for socio-economic and environmental heterogeneity. We thus consider both economic and ecological criteria in the cost-effectiveness analysis of N2K forest contracts.

Our results show that potential income losses from timber production are not accounted for in the payment definition. This payment definition is incoherent since we demonstrate that there is substitutability between biodiversity provision and timber production within the N2000 forest contract framework. These results call for the redefinition of fairer payments, which would make it possible to trigger higher participation, especially from private forest owners who appear to be cost-effective biodiversity producers.

These results, coupled with the literature's findings on the need to adapt existing payment schemes in order to directly tackle future challenges, especially those posed by climate change, (Donald and Evans, 2006; Heller and Zavaleta, 2009) call for a rethinking of the definition of conservation incentives. Thus, **in the second and third chapter of this thesis we leave the framework of N2K contracts. We study the definition of efficient and cost-effective incentive payments in a theoretical and conceptual way, while taking into account the main challenges posed by the definition of incentive payments for biodiversity conservation.**

Chapter 2 explores the possibility of differentiating conservation payments for private landowners when both conservation costs and benefits are heterogeneous and unobservable to the conservation planner. This chapter focuses on the impact of asymmetric information and *adverse selection* on the definition of payments. **In Chapter 3, we investigate the impact of climate change on the definition of cost-effective incentive payments.** In this chapter, we develop an integrated, dynamic and spatially explicit ecological-economic model, and study the relative cost-effectiveness of various payment design options, involving different levels of targeting and differentiation of conservation payments.

We conducted a literature review on the various challenges posed by the definition of

efficient payments for biodiversity conservation and how they have been addressed. This literature review is presented in the remainder of this introductory chapter and explains why it is essential to specifically integrate spatial and dynamic dimensions into the definition of such incentive conservation payments. We then present the contributions of the second and third chapters of this thesis. Our work does not address all the issues presented in the following literature review. The consideration of the stochasticity of conservation benefits in the payment definition as well as the question of the optimal duration of the contract will, for example, be the subject of future research.

3 Designing incentive payments for biodiversity conservation in agricultural and forest landscapes: challenges, policy design options and shortages

3.1 The relevance of the payment for environmental services concept

The concept of PES was initially defined by Wunder (2005) and then redefined more broadly by the same author in 2015 after a decade of theoretical and applied research on PES, in particular as a result of the observation that many existing PES programs never fully illustrated the initial definition, deemed too restrictive (Wunder, 2015 and see e.g. Tacconi, 2012 and Muradian, 2010). We consider the most recent definition and understand a PES as “a (1) voluntary transaction (2) between service users (3) and service providers (4) for generating offsite services” (Wunder, 2015).

There is a wide variety of PES whose structure and definition depend on the nature of the ES targeted, the framework of the transaction and the relationship between ES users and providers (Engel *et al.*, 2008). However, PES are not relevant for all types of ES but are well-suited to encourage biodiversity conservation. Indeed, this type of instrument is particularly relevant when aiming at encouraging the provision of an ES corresponding to a positive externality or the provision of public goods (Engel *et al.*, 2008). In such a case, the PES aims at making the provision of ES attractive, while in the absence of payment, practices associated with the provision of the ES would make landholders incur opportunity costs (i.e., foregone profits from agriculture or timber production). PES thus illustrate the “provider gets” principle in opposition to the “

polluter pays” principle. One of the key features of PES as underlined by Wunder (2015) is the payment conditionality: under PES schemes the payment is conditional either on the implementation of specific management practices allowing ES provision (action-based or input-based conditionality) or on the effective ES provision itself (result-based or outcome-based conditionality). The vast majority of the PES schemes implemented to date have adopted action-based conditionality rather than result-based conditionality (Engel *et al.*, 2008).

Depending on the transaction framework, two main types of PES can be defined: i) PES for which ES “buyers” are the direct beneficiaries are referred to as “user-financed”, ii) while PES for which a third party centralizes the transaction and acts on behalf of the ES beneficiaries are referred to as “government-financed”. In this latter case, the funding for ES provision is centralized and usually based on the collection of a mandatory user-fee.⁷ Incentive payments for biodiversity conservation generally correspond to the “government-financed”-type PES. In this case, the centralization of the transaction by a third party allows, on the one hand, to avoid free-riding problems from ES users and, on the other hand, to increase PES efficiency because of economies of scale in transaction costs. However, in the case of “government-financed” PES, the third party has a lower incentive to ensure the actual provision of the service (or the implementation of the prescribed measures) by ES providers compared to “user-financed” PES. Compliance can also be more costly and difficult to monitor, which is susceptible to undermine the efficiency of “government-financed” PES schemes (Engel *et al.*, 2008).

Thus PES are well-suited instruments when aiming at encouraging biodiversity conservation. However, the definition of cost-effective incentive payments requires to account for several challenges specific to biodiversity.

3.2 Main challenges for the definition of cost-effective incentive payments

Several literature reviews identify the main issues related to the definition of cost-effective PES and show how these can be accounted for in the payment design. In their reference paper, Engel *et al.* (2008) aim to provide an overview of the issues related to the definition of cost-effective PES in general. Börner *et al.* (2017) build on

⁷For example, the financing of incentive conservation payments under Natura 2000 is provided in part by the Member State and in part by the European Union.

this work and identify key determinants of the effectiveness and cost-effectiveness of PES while building on a decade of theoretical and empirical research. For each specific issue, authors identify relevant payment design ensuring a good performance of the PES scheme. Engel *et al.* (2008) and Börner *et al.* (2017) identify three main phenomena which are susceptible to undermine the cost-effectiveness of PES: i) *adverse selection*, ii) *moral hazard* and iii) lack of additionality. Besides, Börner *et al.* (2017) point out the difficulty of defining a clear causal link between the implementation of specific management measures and the effective ES provision. This issue raises the question of payment conditionality and the element on which conditionality should be based.

Hanley *et al.* (2012) as well as de Vries and Hanley (2016) review the literature related to the definition of incentive payments for biodiversity conservation, which Hanley *et al.* (2012) refer to as the “biodiversity problem”. This “biodiversity problem” has three main characteristics: i) heterogeneity of conservation costs and benefits, ii) *hidden information* about these costs and benefits, and iii) stochasticity of conservation benefits. Authors explain how these characteristics can lead to, among others, the emergence of adverse selection and moral hazard phenomena described by Engel *et al.* (2008) and Börner *et al.* (2017).

In the following sections, we show that the characteristics of the “biodiversity problem” are linked to spatial and dynamic processes which are inherent to biodiversity and have to be accounted for in the definition of efficient incentive conservation payments. We build on the literature reviews conducted by Hanley *et al.* (2012) as well as de Vries and Hanley (2016) and extend their work by showing how the different components of the biodiversity problem have been taken into account in the literature dealing with the definition of conservation payments.

4 Accounting for spatial aspects

4.1 Spatial heterogeneity of conservation costs and benefits

In order to be cost-effective, a PES must be able to trigger sufficient participation from ES providers (Engel *et al.*, 2008). This requires to define a sufficient level of payment, that is, payment must be higher than biodiversity conservation costs incurred by land-

holders. However, conservation costs and benefits generally show spatial heterogeneity (see for instance Armsworth *et al.*, 2012). Heterogeneity of conservation costs is explained on the one hand by the heterogeneity of preferences, management practices and skills regarding biodiversity conservation among landowners. On the other hand, properties' characteristics such as land quality or climate vary spatially, thus inducing spatial heterogeneity of conservation costs. Besides, spatial heterogeneity of soil and climate characteristics at various spatial scales influences habitat suitability according to a species' niche. Consequently, different regions of a same country may show heterogeneous potential benefits for the implementation of species-specific conservation programs.

This spatial heterogeneity of conservation costs calls for the definition of differentiated payments aligned with conservation costs (Börner *et al.*, 2017). Indeed, a uniform payment would lead to the overcompensation of all but the marginal landowner.⁸ Moreover, defining cost-effective conservation payments requires to select the most efficient landowners, that is, those providing the maximum level of biodiversity conservation at a given cost.⁹ This calls for the targeting of conservation payments (Börner *et al.*, 2017).

Three types of targeting rules have been described by the systematic reserve design and the conservation planning literature for the implementation of conservation programs: i) cost-targeting, ii) benefit-targeting and iii) benefit-to-cost-ratio targeting. Results of this literature show that the definition of the optimal targeting requires to account for the spatial heterogeneity of both conservation costs and benefits (Naidoo *et al.*, 2006; Duke *et al.*, 2013). The relative cost-effectiveness performance of a targeting rule, compared to the other two, depends on the relative variability of conservation costs and benefits and their correlation (Babcock, 1997).

Several studies have investigated the efficiency gains brought by the definition of differentiated and/or targeted conservation payments. Wätzold and Drechsler (2005) build on Babcock (1997) and investigate the cost-effectiveness losses induced by the definition of uniform payments for conservation in two regions showing heterogeneity in terms of conservation costs and benefits, when compared to payments differentiated at the re-

⁸The marginal landowner is the one showing conservation costs which equal the conservation payment. All landowners showing lower conservation costs are therefore overcompensated, while landowners showing higher conservation costs do not participate in the conservation program.

⁹Or minimizing cost for the provision of a given level of biodiversity output.

gional scale. They show that uniform payments generally lead to low cost-effectiveness performance. Besides, authors extend Babcock's results by showing that the magnitude of cost-effectiveness losses linked to uniform payments depends not only on the relative variability of costs and benefits and their correlation, but also on the shape of the conservation benefit function (i.e., concavity of convexity with respect to the conserved area). Armsworth *et al.* (2012) empirically compare the cost-effectiveness of five AES schemes for biodiversity conservation involving varying degrees of spatial targeting and differentiation of payments. Authors show that the simplification of the payment definition, in particular the absence of spatial differentiation of payments between regions, leads to significant efficiency losses. Armsworth *et al.* (2012) acknowledge that the definition of more sophisticated payment schemes causes additional transaction costs. However, they show that up to 70% of the initial budget allocated to uniform conservation payments could be spent on transaction costs while still leading to cost-effectiveness gains brought by more sophisticated schemes. Wünscher *et al.* (2008) empirically demonstrate that granting differentiated payments through a benefit-to-cost-ratio targeting process would allow significant efficiency gains for the PSA program in Costa Rica. However, Lewis and Plantinga (2007) compare the cost-effectiveness of targeted and uniform payments aiming at reducing habitat fragmentation and show that, depending on initial conditions in the considered landscape, uniform payments can outperform targeted ones. Finally, Lewis *et al.* (2009) investigate the relative cost-effectiveness of targeted, differentiated and uniform payments schemes aiming, there again, at reducing habitat fragmentation. In this case, authors show that the implementation of differentiated payments can bring significant efficiency gains and that this type of payment can outperform targeted ones, being more sophisticated and restrictive.

4.2 Accounting for hidden information on conservation costs and benefits

The definition of socially efficient differentiated and targeted payments by a conservation agency requires to have a sufficiently accurate knowledge of conservation costs and benefits of landowners. While the conservation agency generally has access to information on the spatial distribution of conservation costs and benefits, the level of costs and benefits associated to the conservation of a specific property is usually unobservable to the conservation planner and belongs to landowners' private information. This corre-

sponds to the problem of *hidden information* or (asymmetric information) described by Hanley *et al.* (2012). Under asymmetric information, a landowner has the incentive to declare an untruthfully high level of conservation costs, which allows her to extract an information rent¹⁰: this corresponds to the phenomenon of *adverse selection*. The emergence of adverse selection problem and the existence of information rents undermine the efficiency of conservation payments.

Based on a literature review, Ferraro (2008) shows how asymmetric information and adverse selection issues can be accounted for in the design of PES. The author identifies two main approaches, namely *procurement auctions* and *screening contracts*. Both approaches rest upon revelation mechanisms and consist in framing the contracting process so as to induce landowners to reveal their hidden information. The implementation of such approaches allows to increase the efficiency of PES programs since it enables the reduction (but not the elimination) of information rents.

Ferraro (2008) and de Vries and Hanley (2016) provide a rich literature review dealing with the design of procurement auction mechanisms to incentivize ES provision. However, in both reviews very few studies specifically deal with biodiversity conservation. When implementing procurement auctions, a buyer invites bids from ES providers for a specific contract and selects the most efficient ES providers (i.e., those making the lowest bids). In their seminal paper, Latacz-Lohmann and Van der Hamsvoort (1997) theoretically demonstrate the higher efficiency of procurement auctions compared to fixed-rate payments for ES provision. Bamière *et al.* (2013) show that implementing an auction scheme for biodiversity conservation allows significant cost-efficiency gains compared to a uniform subsidy when a specific spatial landscape configuration has to be established.

Efficiency gains enabled by procurement auctions stems from the price competition fostered among bidders by the auction scheme: landowners have an incentive not to inflate their bid if they want to win a contract. Therefore efficiency gains brought by an auction scheme greatly depend on the sufficient participation from landowners.

Besides, the performance of auctions is highly sensitive to the context in which they are implemented (e.g., single-shot or repeated auctions, level of information of landowners about the total available budget, pricing rule, etc.). The definition of an optimal pricing

¹⁰Understood as the difference between the payment and the actual conservation cost of the landowner.

rule has been investigated in experimental economic studies (e.g., McKee and Berrens, 2001; Cason and Gangadharan, 2005; Krawczyk et al., 2016) and through agent-based models (Hailu and Thoyer, 2006). The relative performance of various auction types in dynamic and static contexts has also been investigated in experimental economic studies (e.g., Schilizzi and Latacz-Lohmann, 2007; Fooks *et al.*, 2015).

Finally, bids can be ranked according to a benefit-cost ratio rather than according to cost only through targeted auctions. Che (1993) and Latacz-Lohmann and Van der Hamsvoort (1997) theoretically demonstrate that such auctions can be more efficient than those ignoring contract quality (i.e., benefits). Efficiency gains brought by targeted auctions depend on the relative variability of costs and benefits as well as on their correlation (Ferraro, 2003). However, the implementation of such targeted auctions assumes that the regulator has a fairly accurate knowledge of benefits offered by each ES provider. Such an assumption may not be valid for biodiversity conservation.

Designing screening contracts consists in defining a “menu of contracts”, namely a unique contract for each level of hidden information (e.g., conservation costs), called “type”. Based on the knowledge of the distribution of landowner types, contracts are defined so that a rational landowner has no choice but to contract truthfully and reveal her type by choosing the matching contract. The implementation of screening contracts allows to reduce information rents but at the cost of downward distortions in the level of output (i.e., ES) provided by less efficient agents. Downward distortions induced by screening contracts are higher than those related to procurement auctions (Ferraro, 2008).

Several studies have addressed the definition of optimal screening contracts for ES provision under adverse selection and a few of them specifically focus on biodiversity conservation. Moxey *et al.* (1999) model optimal AES contracts for pollution reduction. Authors show that AES contracts based on revelation mechanisms allow efficiency gains compared to individually negotiated contracts but do not achieve full information pollution reduction. Gren (2004) compares the performance of uniform payments and compensation payments based on self-selection mechanisms to incentivize land set-aside on environmental purposes. While her analytical results are indeterminate, results derived from an empirical case study demonstrate that contract-based payments for wetland creation show a better performance than uniform ones when aiming at reducing nitrogen

loads. Canton *et al.* (2009) investigate the interest of a decentralized implementation of AES through a principal-agent model. Authors show that a more geographically disaggregated design of AES improves net benefits derived from AES and leads to an increase in the optimal effort required from farmers.

In the aforementioned studies, asymmetric information solely concerns conservation costs; agents are therefore screened on a cost-basis only. Besides, the principal's unique objective is to maximize the amount of area under conservation at least cost. Conservation benefits are usually considered as observable to the principal and are either contractible or directly stem from the level of contractible effort.

Several studies go beyond the screening of landowners on a cost-basis only by a planner with a single objective. Anthon *et al.* (2010) model optimal biodiversity conservation contracts through a principal-agent model involving both adverse selection and moral hazard. Authors account for the heterogeneity and the unobserved character of landowners' ability to achieve a high level of conservation benefits for their forest through conservation. However, they consider conservation benefits as observable and verifiable. Feng (2007) develop a principal-agent model for the design of AES payments with a dual policy goal: the planner (i.e., principal) seeks not only conservation but also income support for small farms. In this work, conservation costs are unobservable and farm size can also be subject to asymmetric information. However, no interdependence between policy goals is accounted for and principal's gains from contracting are defined through two separate value functions.

Designing screening contracts for biodiversity conservation would, in some cases, require to account for asymmetric information on costs and benefits. A conservation planner could for instance wish to target payments towards properties being highly biodiverse or hosting a specific target species. But performing an *ex ante* monitoring to observe species' presence or the biodiversity value of the property *in situ* could let the planner incur high monitoring costs. **In this context we study the definition of optimal conservation payments for a species-specific conservation program and investigate the possibility to define differentiated conservation contracts while accounting for unobservable and heterogeneous conservation costs and benefits.**

Chapter 2

Designing species-specific conservation contracts in a heterogeneous landscape with unobservable conservation costs and benefits

Working paper in Les Cahiers du LEF

Submitted to Environmental and Resource Economics

In this chapter, we consider a principal-agent framework in which a conservation planner delegates the implementation of conservation measures to a landowner within the framework of a species-specific conservation program. The conservation planner has two interdependent objectives: the provision of habitat suitability (through the amount of conserved land) and effective species-specific protection (through the implementation of conservation measures in properties hosting species individuals). The unobservable conservation cost of the landowner is two-dimensional as the sum of opportunity and protection costs. The absence/presence of the target species in a property is signaled by the level of protection cost of the landowner. We render this conservation problem through a *common-value* model in which the principals' preference for both types of conservation benefits is explicitly taken into account.

We contribute to the literature applying contract theory to PES design by accounting for the unobservable character of conservation benefits as well as for the multidimensionality of asymmetric information, which has been rarely done in this literature (see Feng, 2007). This analytical framework is more comprehensive and better suited to the definition of optimal contracts for species-specific conservation.

We analytically demonstrate the possibility to define differentiated conservation payments despite a *non-responsiveness* situation, known to usually lead to bunching equilibria. Results of numerical landscape-scale simulations show that contracts derived from a common-value model can perform better than those derived from a classic adverse selection model. Indeed, we find differentiated contracts, which are closer to first-best ones and show interesting cost-effectiveness performances.

Finally, in this work we implicitly account for the spatial dimension of the heterogeneity in conservation benefits by investigating the impact of a variation in the target species presence probability on the performance of contracts. This allows us to make

recommendations on the type of contract to be implemented based on the considered region within the species range.

4.3 Specific species requirements, spatial externalities and spatial dependency of conservation benefits

When aiming at biodiversity conservation at the landscape scale, the realization of conservation benefits can in some cases require the establishment of a specific spatial configuration of conserved properties (i.e., optimal land-use pattern). This optimal land-use pattern is generally dictated by spatial processes underlying the dynamic of the target species. Some species being for instance sensitive to fragmentation and edge effects would require the establishment of agglomerated conserved areas, whereas others on the contrary would do best in fragmented landscapes. A rich body of economic literature has integrated such ecological constraints into the definition of conservation payments.

Based on a theoretical model combined with an econometric model and GIS-based simulations conducted at the landscape scale, Lewis *et al.* (2009) compare the performance of three types of conservation payments aiming at reducing habitat fragmentation. Conservation results derived from each payment type are assessed through an index calculating habitat fragmentation at the landscape scale.

Bamière *et al.* (2013) compare the performance of three types of incentive conservation payments when the optimal landscape pattern is a random mosaic. Authors consider a uniform subsidy, an auction scheme and an agglomeration malus. They model the landowner's behavior at the farm scale and analyze the resulting landscape configuration of conserved land at the regional level through an indicator. This indicator allows to assess the suitability of the emerging landscape pattern with respect to species requirements. Authors show that the agglomeration bonus is more cost-efficient than a uniform subsidy but cannot be ranked compared to the auction scheme.

Smith and Shogren (2002) compare the performance of two types of screening contracts – *ex ante* and *ex post* contracts – for biodiversity conservation in a multi-landowner context. In this study, conservation results derived from contracts partly depend on

the achievement of a “minimum-area” constraint: the amount of conserved land should be greater or equal to this constraint to ensure species’ survival. In the case of *ex ante* contracts, a contract offered to a specific landowner type does not depend on contracts aimed at the other types. In the case of *ex post* contracts, conservation contracts are interdependent. Authors show that *ex ante* contracts never implement first-best allocations and can lead to an excess of land set aside. First-best allocations can be implemented under *ex post* contracts only and this type of contracts shows a higher cost-effectiveness performance compared to the *ex ante* type.

Parkhurst et al (2002) introduce the concept of agglomeration bonus (AB) consisting in rewarding – on top of uniform compensatory payments – those landowners in a group who manage to coordinate and conserve contiguous parcels. The literature in experimental economics has largely appropriated this concept and tested its applicability and performance in different contexts. Results from this literature have showed that the magnitude of efficiency gains brought by an AB vary according to the context in which payments are implemented. Parkhurst *et al.* (2002) show that allowing landowners to communicate with each other in the group increases efficiency gains brought by the AB. Besides they show that the implementation of long term AB contracts without communication (with communication) lowers (increases) efficiency gains brought by the AB. In addition, Parkhurst and Shogren (2007) show that efficiency gains brought by the AB vary according to the spatial structure of the targeted reserve. Finally, the efficiency gains due to the AB also vary according to the landowners’ experience in contracting (Parkhurst and Shogren, 2007), the form and size of the landowners network (Banerjee *et al.*, 2012) as well as the structure of the information given to landowners (Banerjee *et al.*, 2014).

Drechsler *et al.* (2010) propose an agglomeration payment (AP) differing from the AB in the following way: in this case the whole conservation payment is made to landowners if and only if the resulting conservation network shows a level of spatial connectivity being greater or equal than a threshold value defined by the conservation planner. The connectivity of the resulting conservation network is assessed through an ecological function. The design of an AP is based on a payment level and a threshold value for landscape connectivity. Since the payment is uniform, meeting the connec-

tivity threshold requires coordination and side-payments between landowners. Authors compare the performance of the AP (conditional) and an unconditional uniform conservation payment on a conceptual level. They show that implementing an AP enables significant efficiency gains compared to a uniform payment, especially for low levels of available conservation budget. Wätzold and Drechsler (2014) build on Drechsler *et al.* (2010) and conceptually compare the cost-effectiveness and budget efficiency of three types of conservation payments involving i) uniform payments, ii) an AB iii) or an AP. They show that the AB is always outperformed either by uniform payments or by the AP (but never by both). Authors show that the cost-effectiveness and budget efficiency gains brought by the AP are influenced by the spatial correlation of conservation costs as well as by the characteristics of the target species (i.e., dispersal ability).

Thus, accounting for spatial dimensions and processes related to biodiversity can significantly influence the design of an optimal conservation payment. In the following section, we show that accounting for stochasticity of conservation benefits as well as for temporal issues can also significantly influence the choice of the “right type” of conservation payment.

5 Accounting for stochasticity of conservation benefits and temporal issues

5.1 Stochasticity of conservation results: input-based vs. output-based payments

As underlined by Wunder (2015), conditionality is one of the key features of PES. As mentioned earlier, two types of conditionality can be applied: i) strict conditionality leading to the definition of output- or result-based payments or (ii) “proxy conditionality” leading to the definition of input based-payments. The question about which one of these two types of payment design is optimal has been increasingly debated in the conservation literature over the last decade (Gibbons *et al.*, 2011; Derissen and Quaas, 2013).

As underlined by Hanley *et al.* (2012) the design of incentive payments for conservation has to take into consideration that results derived from the implementation of conser-

vation measures are stochastic from the landowner's perspective. Indeed, conservation results are only partly determined by these conservation actions and can be influenced by external factors such as extreme events. Based on a literature review, Maron *et al.* (2012) show that ecological restoration for biodiversity offsets is subject to three main limits being: i) poor measurability and ii) uncertainty of results derived from restoration actions as well as iii) time lags affecting these results. In this context, it seems therefore legitimate to base conditionality on compliance with the implementation of prescribed conservation measures. This input-based conditionality has been adopted in the majority of existing PES schemes. It has been demonstrated that input-based payments would perform well in a case in which the conservation agency has a perfect knowledge of the impact of the implementation of conservation actions by landowners on the provision of the targeted ES (Gibbons *et al.*, 2011; Derissen and Quaas, 2013; White and Hanley, 2016). However, when the actual implementation of certain measures being crucial to achieving the conservation objective is unobservable to the conservation planner, implementing output-based payments can be more efficient. Indeed, in this case there is no incentive for a landowner to exert much effort for biodiversity conservation, which is likely to lead to a poor conservation result. This corresponds to the *moral hazard* phenomenon. Several studies have investigated the relative performance of input-based and output-based payments for ES provision and especially for biodiversity conservation.

Output-based payments have been highlighted as flexible instruments allowing to trigger innovation from participating landowners (Matzdorf and Lorenz, 2010; Gibbons, 2011). They would also be particularly helpful in a context of asymmetric information. In their study, Anthon *et al.* (2010) show that offering a two-part contract, involving an unconditional base-payment and a result-based payment, to landowners likely to provide high conservation benefits is optimal under moral hazard. A similar result is obtained by White and Hanley (2016) who study the definition of optimal incentive payments for ES provision under hidden information about provision costs and *hidden action*. Derissen and Quaas (2013) investigate through a principal-agent model the influence of asymmetric information and uncertainty about the definition of optimal incentive payments for ES provision. In this work, authors consider that ES provision is subject to environmental uncertainty; besides authors consider that the marginal productivity of management actions for ES provision is unobservable to the conservation planner

and belongs to landowners' private information. In this context, Derissen and Quaas (2013) show that implementing a combined scheme, involving an input-based and a result-based payment, always increases social welfare compared to both pure result-based and pure input-based payments for a risk neutral regulator. Besides, authors show that the result-based part of the payment increases with the degree of asymmetric information. However a high stochasticity of conservation benefits is likely to undermine the performance of result-based payments: Derissen and Quaas (2013) show that the result-based (input-based) part of the optimal incentive decreases (increases) with the degree of environmental uncertainty. This result is in line with results highlighted by Zabel and Roe (2009) in their literature review.

Moreover, output-based payments can be perceived as riskier by participating landowners, which can lead the latter to require a risk premium (Zabel and Roe, 2009). This would result in higher levels of demanded payments and thus higher program's costs. In this respect, both Zabel and Roe (2009) and Derissen and Quaas (2013) show that the result-based part of the payment should decrease with increasing landowner's level of risk aversion.

5.2 Optimal timing and contract length

The influence of temporal dynamics underlying the provision of ES has received less attention than spatial dimension and processes in the literature. Yet, these temporal dynamics are also likely to have a significant influence on the definition of optimal incentives. As previously pointed out, the realization of the benefits stemming from the implementation of conservation actions is affected by time lags. Besides, depending on the objective, conservation measures must be continuously implemented for a sufficiently long period of time to actually deliver conservation benefits. Nevertheless, a few studies have investigated this question.

Gulati and Vercaemmen (2005) investigate the determinants of optimal contract length for carbon sequestration and show that – consequent to the decreasing marginal benefits but increasing opportunity costs of the contract over time – the optimal contract should be finite. Lennox and Armsworth (2011) examine how the optimal contract length is affected by uncertainty about future availability and ecological condition of eligible sites. Ando and Chen (2011) show how the turnover rate, the biological benefit function and

the average income of private land influence the optimal length of incentive contract for ES provision. Authors show that the speed at which environmental benefits mature significantly affects the optimal contract length. Juutinen *et al.* (2012) study the influence of forest stand characteristics and habitat requirements of target species on the optimal length of conservation contracts in boreal forests. Authors show that a cost-effective policy involves both short- and long-term contracts, and that optimal contract length increases with increasing conservation objectives. Juutinen *et al.* (2014) build on this work and examine how transaction costs and conservation budget (type and size) influence the optimal contract length for biodiversity conservation. Shah and Ando (2016) investigate the definition of efficient incentives for ES provision in a dynamic framework with uncertainty about conservation and agricultural returns. Authors compare the performance of temporary and permanent contracts through a real options framework and show that temporary contracts outperform permanent contracts, especially in a context of high uncertainty and low discount rate.

6 Integrated ecological-economic modeling

6.1 Contributions of integrated ecological-economic modeling

The studies mentioned in sections 4 and 5 show the need to account for spatial and dynamic dimensions of the biodiversity problem in the definition of efficient conservation incentives. However, in the aforementioned studies, biodiversity and related spatial and dynamic processes are generally accounted for as a constraint, against which conservation results derived from economic incentives are assessed. Interdisciplinary ecological-economic approaches go a step further: such approaches allow to account for both economic and ecological aspects of the conservation problem with a sufficient level of complexity in the definition of incentives. This strand of literature has greatly expanded over the last decades; it combines the knowledge, gathered by economists, of agents' (i.e., landowners') behavior with knowledge, gathered by ecologists, of spatial and dynamic processes underlying species' behavior.

Integrated ecological-economic models generally involve three basic steps described below and summarized in Figure 1:

1. Landowner's behavior model: Landowner's behavior is usually modelled at the

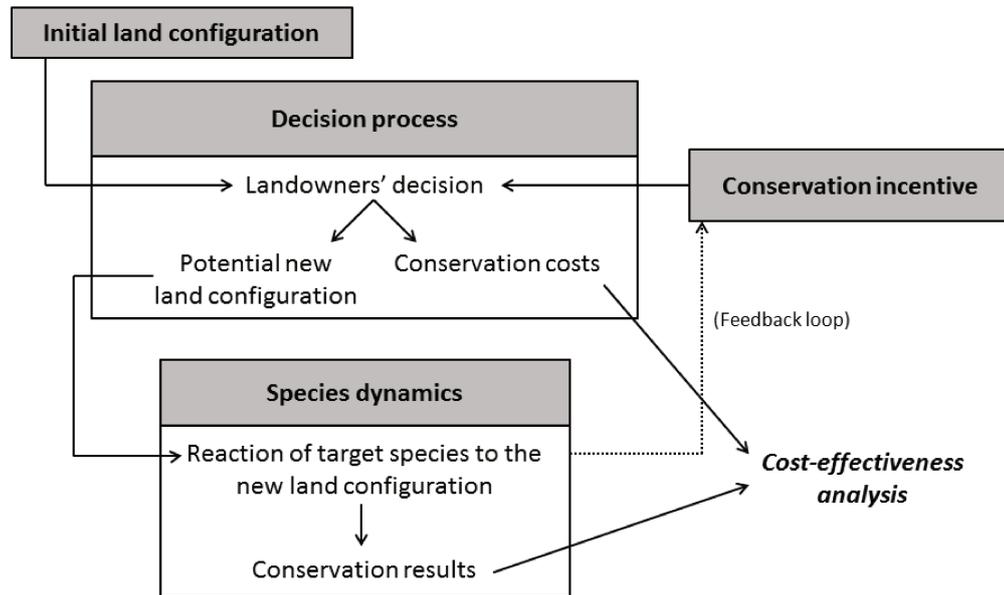
parcel or property scale. This model allows to determine landowner's reaction to various conservation incentives. Reactions of landowners to these incentives lead to the emergence of a specific land-use pattern.

2. Ecological model: The emerging land-use pattern is characterized in terms of conservation costs and by various landscape metrics: connectivity, total size of the conserved area, turnover rate etc. Based on these metrics, ecological models determine the reaction of the target species to this land-use pattern - usually at the landscape scale - and thus determine conservation results.
3. Analysis of incentives' performance: The determination of conservation costs and results linked to each type of incentives allows to perform a (comparative) cost-effectiveness analysis.

Undertaking an ecological-economic approach allows on the one hand to stress existing ecological-economic trade-offs (Doyen *et al.*, 2013). For instance, Barraquand and Martinet (2011) develop a spatially explicit ecological-economic model and study existing trade-offs between agricultural production and biodiversity conservation in a dynamic landscape under uncertainty (i.e., crop prices' volatility). Authors determine the production possibility frontier of the dynamic landscape and show the possibility to generate ecological outcome with small production losses. Mouysset *et al.* (2015) develop a multiscale bio-economic model and assess the performance of optimal various agricultural policies (i.e., taxes and subsidies) based on their public, private and social costs when applied in metropolitan France. Authors demonstrate the existence of trade-offs between economic and biodiversity objectives from a private cost perspective but show the possibility of bio-economic synergies from a social cost perspective.

On the other hand, ecological-economic models enable the definition of fine-tuned conservation incentives since they allow to understand the influence of economic and ecological parameters on the performance of conservation incentives. For example, Drechsler *et al.* (2007) study the design of cost-effective conservation instruments for multi-species conservation in a spatially heterogeneous landscape (in terms of opportunity cost of conservation) with habitat dynamics. Authors show that the degree of substitutability between target species for the regulator as well as species characteristics (e.g., dispersal ability) influences the cost-effectiveness of the various considered conservation instru-

Figure 1: Basic steps of integrated ecological-economic modeling - Adapted from Hartig and Drechsler (2010)



ments. Wätzold and Drechsler (2014) show how species dispersal ability as well as the degree of spatial correlation of costs influences the relative cost-effectiveness performance of AP, AB and homogeneous payment. Besides, applying such ecological-economic models allows to formulate specific recommendations for the implementation of conservation incentives: Johst *et al.* (2002) determine the efficient spatial and temporal allocation of conservation incentives for White Stork conservation based on conservation cost data for Saxony (Germany). Finally, Wätzold *et al.* (2016) assess the cost-effectiveness of existing AES for grassland conservation in Saxony. Authors also propose non-existing AES and estimate the potential ecological impacts and cost-effectiveness of the latter. Wätzold *et al.* (2016) detect that substantial cost-effectiveness improvements could be made for existing AES: authors show that more accurately accounting for temporal species-specific requirements in the implementation of AES would enable to achieve the same conservation results with a 28% lower conservation cost.

6.2 Integrated ecological-economic modeling and the design of incentive tools for biodiversity conservation under climate change

Existing ecological-economic studies have thus investigated the design of efficient conservation incentives, especially in dynamic landscapes. However, to our knowledge none of them takes into consideration climate-induced landscape dynamics and investigates the definition of incentives for biodiversity conservation under climate change. Yet, climate change is expected to be one of the key threats for biodiversity conservation in this century (Sala *et al.* 2000) and, despite of the interest underlined by the literature of existing payment for ecosystem services (PES) schemes for the establishment of “climate-proof” conservation networks (Donald and Evans, 2006; Heller and Zavaleta, 2009), these existing payments schemes need to be reformed in order to directly address challenges posed by climate change (Donald and Evans, 2006).

Several studies have investigated the possibility to establish dispersal corridors or priority areas for conservation under climate change while accounting for conservation costs and climate-induced land-use change with varied levels of complexity (Williams *et al.*, 2005; Vos *et al.*, 2008; Faleiro *et al.*, 2013 and Tainio *et al.*, 2014). But, in these studies economic aspects of the conservation problem are accounted for in a limited manner. In contrast, Leroux and Whitten (2014) and Mallory and Ando (2014) considered the impact of climate change on conservation policies from an economic perspective. However, they addressed the ranking of competing conservation projects (Leroux and Whitten, 2014) and the prioritization of conservation areas (Mallory and Ando, 2014) but did not investigate the policy instrument of conservation payments. Thus, research on biodiversity conservation under climate change has largely remained confined to the conservation planning literature and ignored incentive-based conservation policy instruments (but see Ay *et al.*, 2014).

In the third chapter of this thesis, we build on previous research about the cost-effectiveness of conservation payments in integrated ecological-economic modelling and investigate the definition of conservation payments under climate change.

Chapter 3

Cost-effectiveness of conservation payment schemes under climate change

Working paper in Les Cahiers du LEF

We develop a spatially explicit ecological-economic model in a landscape whose dynamics is driven by climate change. We investigate the cost-effectiveness of various payment design options involving varying degrees of payments' differentiation and targeting, while considering the impact of changes in key economic and ecological parameters.

We provide the first comparative cost-effectiveness analysis of conservation payment designs in a changing climate on a conceptual level. The modeling of targeted payments requires the consideration of feedback loops between the economic and ecological models that we integrate into our work.

Our results demonstrate the significant cost-effectiveness gains enabled by payments' differentiation and targeting for biodiversity. Moreover, we show that connectivity/area trade-offs, highlighted in the context of dynamic landscapes, are also observed under climate change. The cost-effectiveness performance of targeted payments compared to untargeted differentiated payments increases with a decreasing species dispersal ability but decreases with decreasing climate stability in the landscape.

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Chapter I

Assessing the cost-effectiveness of a biodiversity conservation policy: A bio-econometric analysis of Natura 2000 contracts in forest

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Abstract

In France, the implementation of the EU biodiversity conservation policy within the Natura 2000 network combines regulatory tools and voluntary contracting. In this chapter, we empirically assess the cost-effectiveness of Natura 2000 contracts in forest areas. We simultaneously estimate a cost function for biodiversity conservation and the production set of biodiversity output and timber, while controlling for conservation measures. We show strong substitutability between biodiversity conservation and timber production. Estimate results on the cost-elasticity of biodiversity conservation also suggest the possibility of more ecologically ambitious contracts with lower average costs. Results also show that public owners are able to bear higher opportunity costs than private owners. Our findings may help to formulate policy recommendations in terms of contracts' targeting, likely to enhance the cost-effectiveness of the incentive scheme.

Keywords: Forest, Natura 2000, biodiversity conservation, biodiversity index, cost-effectiveness, payment for ecosystem services

1 Introduction

The European Union (EU) Natura 2000 (N2K) policy aims at long-term conservation of species and habitats of Community interest on EU territory. Based on the Birds and Habitats Directives, French administration chose to manage N2K sites on a participative basis by allowing local agents to voluntarily contract for biodiversity conservation.

Biodiversity is a public good and a positive externality deriving from landowners' activities. It is thus provided in a sub-optimal way in spite of its significant social value. By signing an N2K contract, landholders commit themselves to implement biodiversity conservation measures on their land, aiming at maintaining and/or enhancing the initial level of biodiversity, and receive an input-based payment in exchange.¹ The aim of payments for N2K contracts is to trigger biodiversity conservation by landholders in order to avoid degradation of the public good (i.e., biodiversity loss). Thus, the output of the contract is to protect the value of the public good which is already there as well as to address conservation needs. In this respect, N2K contracts may be considered as a Payment for Environmental Services (PES) scheme (Wunder, 2005; Tacconi, 2012).

In spite of the growing success and implementation of PES over the last decades, only a few studies offer an empirical *ex post* evaluation of biodiversity conservation payments (Ferraro and Pattanayak, 2006; Wunder *et al.*, 2008; Pattanayak *et al.*, 2010). There has been particularly few studies assessing the cost-effectiveness of biodiversity conservation instruments in the EU, and especially within the N2K framework (Wätzold and Schwerdtner, 2005). Kleijn and Sutherland (2003) investigate the ecological effectiveness of agri-environment schemes that have been implemented on EU territory. However, the study does not take any economic aspects into consideration. Matzdorf and Lorenz (2010) undertake the first empirical cost-effectiveness evaluation of result-oriented agri-environment measures (roAEMs) in Germany and show the higher cost-effectiveness of the latter, but their evaluation is mainly qualitative and does not consider ecological aspects. Finally, Wätzold *et al.* (2010) conceptually and qualitatively assessed the cost-effectiveness of managing N2K sites but did not specifically address payments for biodiversity conservation.

¹In addition to N2K contracts, regulatory instruments have been implemented within N2K sites. Local landholders can also voluntarily sign the N2K site's charter and implement "good management practices" on their land. They receive no payment but a tax refund in exchange.

To our knowledge, very few studies quantitatively assess the implementation of the N2K policy in forests areas. Jacobsen *et al.* (2013) evaluate economic consequences of restrictions on regeneration intensity and setting aside of some tree stands as recommended by N2K policy. This valuation is conducted with an approach based on the foregone net present income. Rosenkranz *et al.* (2014) determine income losses due to the implementation of conservation measures within the N2K framework for individual forest enterprises. They take individual characteristics of forest enterprises (e.g., management practices, level of environmental and production objectives, structure of forest stands, etc.) into account when defining opportunity cost of conservation. However, this study focuses on the impact of the implementation of the Habitats Directive on a single habitat type and does not consider ecological aspects. Anthon *et al.* (2010) study the design of Danish and French N2K forest contracts. Referring to contract theory, authors conclude that the definition of incentives within French policy is inaccurate, probably undermining the efficiency of the latter. However, this study does not include any empirical analysis of implemented contracts. Finally, Hesmatol-Vaezin *et al.* (2014) investigate (*ex post*) cost-effectiveness of biodiversity conservation using difference-in-difference estimators and inverse dose-response models. This pilot study shows that biodiversity conservation measures implemented within N2K French forest contracts were “significantly effective in promoting a degree of biodiversity conservation”. However, authors conduct their analysis on 28 N2K forest contracts and three types of conservation measures only.

The goal of our study is to assess the cost-effectiveness of the EU environmental policy N2K, which has been implemented in the form of contracts in France. We undertake an *ex ante* approach: we use data related to targeted species and habitats at the individual contract level and assess the cost-effectiveness of N2K contracts at conserving this preexisting level of biodiversity. Our contribution is twofold. First, while conservation schemes are usually qualitatively and globally assessed, we estimate a cost function of biodiversity conservation from individual contract data, controlling for socio-economic and environmental heterogeneity. Second, we use a biodiversity index to model and estimate the production set of biodiversity and timber, accounting for heterogeneity of biodiversity conservation measures. This makes it possible to consider both economic and ecological criteria in the cost-effectiveness analysis of N2K contracts.

In this chapter, we confirm that potential income losses from timber production are not taken into consideration in the payment's definition. This definition is inaccurate since we show that there is substitutability between biodiversity provision and timber production within the N2K forest contract framework. Redefining fairer payments would make it possible to encourage participation from private forest owners who appear to be cost-effective biodiversity producers.

The remainder of this chapter is structured as follows. In section 2, we give a brief overview of the conservation instruments used in N2K in France. In Section 3, we present our economic approach, the method chosen for the development of the biodiversity index, and data. Results are presented in Section 4. In the last section, we discuss policy recommendations derived from our results.

2 Biodiversity conservation in French forests under the N2K policy

The legal framework for the implementation of this policy has been established by the Birds and Habitats Directives. The N2K network is the cornerstone of this conservation policy; it consists of a set of terrestrial and marine protected natural areas designated by the Member States on the basis of the presence of species and habitats of Community interest.

The French N2K network is made up of 1,766 sites, of which 927 comprise forest ecosystems. Biodiversity conservation within the French N2k network is based, on the one hand on, regulatory instruments — strict protection systems and the mandatory assessment of implications for N2K sites of onsite projects and development plans — and, on the other hand, on incentive instruments such as N2K Charter and N2K contracts.

A strict protection system must be implemented for all species listed in the Annex IV of the Habitat Directive and this in all EU Member States. The implementation of these strict protection systems is governed by Articles 12, 13 and 16 of the Habitats Directive. Strict protection applies both to individuals of the species as well as to its habitat, resting and breeding sites. Strict protection systems have to be implemented not only within N2K sites but also throughout the EU territory.

The N2K impact assessment is governed by Article 6 of the Habitats Directive. This regulatory instrument requires that any development project planned within a N2K site that is not linked to the management of the site and likely to have a significant impact on biodiversity should be assessed. This assessment is mandatory for any public or private project and must prove that the implementation of the project would not conflict with onsite conservation objectives.

The Habitats and Birds Directives decrees that Member States implement and enforce these regulatory instruments, but allow greater freedom in the management of N2K sites. The French administration has chosen to combine the implementation of these regulatory tools with incentive instruments allowing two levels of commitment of private owners and public managers to conserve biodiversity. An owner/manager can choose to voluntarily comply with the site-specific N2K Charter and implement good management practices on her property for a five years or ten years period. In this case, no payment is made to the owner but the latter is exempt from property taxes for the area concerned by the implementation of the Charter.

An owner can also choose to enroll in a N2K contract. In this case, the landowner commits to implementing conservation measures in her property that significantly contribute to onsite biodiversity protection. The duration of the contract ranges from five to thirty years. For each N2K site, a specific management plan clarifies which conservation measures can be implemented and defines corresponding technical specifications. Landowners enrolled in a N2K contract are granted a two-part input-based payment and are also exempt from property taxes. Four types of N2K contracts are currently implemented in France: “neither agricultural nor forestry” contracts, agri-environment schemes (AES), marine N2K contracts and forest N2K contracts. In our work, we focus on the implementation of N2K contracts in forests.

3 Empirical strategy

The aim of our empirical study is to assess the cost-effectiveness of N2K forest contracts in France, both in economic and ecological terms. The latter is based on the study and the estimation of a cost function related to the implementation of different conservation measures. Contract’s output is defined by the environmental goal, that is the preser-

vation of the initial level of biodiversity in the intervention area, which corresponds to avoiding of the degradation of targeted species' and habitats' conservation status. Thus, we construct a proper output quantifying this initial level of biodiversity, by accounting for targeted habitats and species. We link this biodiversity output with other forest activities and estimate the production set of biodiversity and timber, depending on actions implemented for biodiversity conservation.

3.1 A variable cost function of biodiversity conservation

Following Armsworth *et al.* (2012), we consider N2K contract participants as biodiversity producers. We choose to study costs associated with biodiversity conservation in order to assess the cost-effectiveness of N2K forest contracts. We assume that costs incurred in N2K contracts are the minimum expenses required for biodiversity conservation given variable input prices and quasi-fixed input.

Forest owners are also producers of other ecosystem services provided by the forest, and particularly of timber. Production of different outputs incurs different costs, some of which are complementary and others substitutable. In all cases, in a context of multifunctionality of forests, modeling a multiproduction cost function would be well adapted. However, available cost data are related to conservation contracts and exclude those costs incurred by the management of the forest to other purposes. This is why we focus our study on biodiversity conservation alone, considering a separable cost function in output. Nevertheless, we still have to test this hypothesis, and in particular, possible interactions between timber production and conservation measures, as well as estimate potential effects on contract costs. Moreover, we model and estimate the production relationship between biodiversity output and timber production, simultaneously to the conservation cost function, to account for the "ecological technology" in the assessment of N2K contracts' cost-effectiveness.

We consider a cost function for the participants in N2K policy signing a contract allowing to provide an output that is the conservation of biodiversity (preexisting in the intervention area), and referred to as y . Within the framework of biodiversity conservation and given the lack of specific information, we cannot observe either the use of "classical" inputs such as labor, energy and materials, or the prices of these

inputs. This is not prejudicial since we do not study substitution effects for inputs.² However, our cost function estimation integrates different variables such as land value (see Section 2.3), to proxy quasi-fixed inputs. These variables are referred to as x . Finally, several additional hedonic variables allow us to precisely define the context of biodiversity production (e.g., type of landholder benefiting from the contract, mean slope and altitude of the property). These hedonic variables are summarized in the vector z . The variable cost function for biodiversity conservation can thus be written as:

$$C(y, x, z, \varepsilon_C), \quad (\text{I.1})$$

where ε_C corresponds to the error term accounting for unobservable variables.

3.2 A biodiversity index to quantify the level of biodiversity output

Two options are available for quantifying the biodiversity output of a contract: (1) the *ex post* approach; and (2) the *ex ante* approach. The former consists in directly measuring the ecological impact of conservation measures on biodiversity (i.e., species and habitats of Community interest). Nevertheless, Yon *et al.* (2003) and Marage (2013) have shown the poor quality of the French monitoring system. Consequently, we do not evaluate the cost-effectiveness of payments for N2K contracts through this type of *ex post* analysis. In this study, we undertake an *ex ante* approach: we use data related to targeted species *and* habitats at the individual contract level *before* conservation measures have been implemented to calculate a two-part contract-specific biodiversity index value quantifying this level of preexisting biodiversity targeted by the contract. As we explained previously, the aim of N2K contracts is to encourage landholders to conserve existing biodiversity in their property in order to avoid biodiversity erosion, rather than to systematically enhance the level of biodiversity. Thus, an *ex ante* approach is valid and allows quantifying the level of biodiversity output.

The index we use stems from the Systematic Reserve Design and Site Prioritization literature. In this literature, indexes traditionally measure the (non-monetary) “biodiversity value” of conservation sites. We use a biodiversity index, that was developed by

²This amounts to assuming that the cost function is separable between the output and the input prices.

Rey Benayas and de la Montaña (2003) and used by Abellán *et al.* (2005):

$$\text{Species Biodiversity Index } (BI_s) = \sum_{i=1}^S \left(\frac{1}{N_i}\right) * V_{ij} \quad (\text{I.2})$$

This index accounts for three of the main criteria used when building a biodiversity index: species richness, through the sum $\sum_{i=1}^S$, where S is the total number of targeted species in the contract j ; the j site-specific vulnerability of species i through the term V_{ij} ; and species' rarity through the term $\frac{1}{N_i}$.³

We did not base our biodiversity index on specific indicator taxa⁴, since the designation of N2K sites requires an initial inventory of all habitats and species of Community interest present on the site, as well as an initial monitoring of their conservation status.

Finally, habitats (and especially habitat vulnerability) should also be considered when building biodiversity indexes and establishing conservation priorities, since they influence rarity (Gauthier *et al.*, 2010). Thus, we consider the following index for habitats:

$$\text{Habitats Biodiversity Index } (BI_h) = \sum_{h=1}^H \left(\frac{1}{N_h}\right) * V_{hj} \quad (\text{I.3})$$

The habitat biodiversity index is built the same way as the species biodiversity index, where H is the total number of targeted habitats in the contract. Our final biodiversity index is then the sum of the two sub-indexes for species and habitats:

$$\text{Biodiversity Index } (BI) = BI_s + BI_h \quad (\text{I.4})$$

We directly draw richness and rarity criteria from naturalist inventories of the official N2K database. However, we have to develop our own scoring method in order to calculate a vulnerability score and determine vulnerability terms (V_{ij} and V_{hj}). We consider three criteria: (1) Red List Status of species i (e.g., Simaika and Samways, 2009; Rey Benayas and de la Montaña, 2003), according to the French National IUCN Red List of endangered species, providing a score we call R_i ;⁵ (2) Priority Status of species i (resp.

³Rarity is calculated at the national scale and N_i corresponds to the number of N2K sites in France where species i appears.

⁴“Indicator taxa are species of higher taxonomic groups whose parameters, such as density, presence or absence, or infant survivorship, are used as proxy measures of ecosystem conditions” (Hilty and Merenlender (2000)). These species are usually regularly monitored and well documented.

⁵The scoring method adopted for the R_i score is adapted from Simaika and Samways (2009).

habitat h) of Community interest as defined by the EU Habitats and Birds Directives (e.g., Gauthier *et al.*, 2013), providing a score we call P_i ; and (3) Conservation Status of species i (resp. habitat h) at the N2K site scale (e.g., Gauthier *et al.*, 2013), providing a score we call C_{ij} .

The vulnerability of species i (V_{ij}), on the N2K site j is equal to the arithmetic mean of three scores: R_i , P_i and C_{ij} . In the same way, vulnerability of habitat h corresponds to the arithmetic mean of two scores: P_h and C_{hj} .⁶ Table I.1 describes the scoring method. Considering species and habitat rarity, as well as their conservation status in the definition of our biodiversity index allows us to accurately render prioritization criteria expressed in official N2K contract design.⁷

Table I.1: Determinants of species vulnerability

Vulnerability (V_{ij}) determinants for species i in the N2K site j					
National IUCN Red List		Habitats Directive: List of priority species		French Standard Data Forms	
IUCN Category ^a	Corresponding R_i score	Category	Corresponding P_i score	Conservation Status	Corresponding C_{ij} score
CR, EN	4	Priority species: YES	4	D	4
VU, NT	3			C	3
LC	2	Priority species: NO	1	B	2
Not considered	1			A	1

^a IUCN Categories : CR (Critically Endangered), EN (Endangered), VU (Vulnerable), NT (Near Threatened), LC (Least Concern)

We calculate a contract-specific value of our biodiversity index (BI) for each contract in our sample. The index then takes its values in the $]0;3[$ interval and quantifies the biodiversity output in two ways. First, the index quantifies the “biodiversity value” (increasing with species richness and rarity) that is conserved by each contract through the implementation of conservation measures. Second, by accounting for the vulnerability of targeted species and habitats, our index makes it possible to quantify the level of threat and thus the level of conservation needs addressed by the contract.

We also calculate a site-specific value of our biodiversity index for each N2K forest site in France. This allows us to assess the quality of targeting for the implementation of

⁶There is no existing National IUCN Red List for habitats as of this time. We therefore measure the vulnerability of habitats of Community interest based on two criteria only: priority and conservation status.

⁷The ministerial memorandum DGPAAT/SDDRC/C2012- 3047 (MEDDE, 2012) defines prioritization rules in order to target payments towards the conservation of (i) species and/or habitats with poor conservation status at the scale of the biogeographical region; (ii) priority species and/or priority habitats of Community interest; (iii) rare species and/or habitats; and (iv) species and/or habitats with poor conservation status at the Natura 2000 site scale.

N2K contracts in terms of ecological benefits.

3.3 Bio-econometric model

In order to avoid constraining the representation of the process of biodiversity conservation, we specify the cost function with a flexible functional form. We estimate a translog cost function (Christensen *et al.*, 1971), as used in industrial organization. It is a local approximation, so that we choose the average of each variable (expressed in the logarithmic form) as a reference point. Consequently, first-order coefficients can be directly interpreted as cost-elasticity terms. We have the following expression of the cost function:

$$\begin{aligned}
\ln(C) = & \alpha_0 + \alpha_{BI}\ln(BI) + \alpha_{LV}\ln(LV) + \alpha_A\ln(A) + \sum_k \beta_k Z_k \\
& + \frac{1}{2}\alpha_{BIBI'}[\ln(BI)]^2 + \frac{1}{2}\sum_k \sum_{k'} \beta_{kk'} Z_k Z_{k'} + \frac{1}{2}\alpha_{LVLV'}[\ln(LV)]^2 \\
& + \frac{1}{2}\alpha_{AA'}[\ln(A)]^2 + \sum_k \gamma_{BIk}\ln(BI)Z_k + \gamma_{BILV}\ln(BI)\ln(LV) \\
& + \gamma_{BIA}\ln(BI)\ln(A) + \delta_{LVA}\ln(LV)\ln(A) + \sum_k \delta_{LVk}\ln(LV)Z_k \\
& + \sum_k \eta_{Ak}\ln(A)Z_k + \epsilon_C,
\end{aligned} \tag{I.5}$$

where LV represents the variable LANDVALUE, and A and represents the size of the forest area that is enrolled for the implementation of conservation measures. The Z_k vector is a vector of dummy variables that provides information about the type of contractor. Parameters $(\alpha_0, \alpha_{BI}, \alpha_{LV}, \alpha_A, \beta_k, \alpha_{BIBI'}, \beta_{kk'}, \alpha_{LVLV'}, \alpha_{AA'}, \gamma_{BIk}, \gamma_{BILV}, \gamma_{BIA}, \delta_{LVk}, \delta_{LVA}, \eta_{Ak})$ have to be estimated.

We also model the “ecological technology” by a production function linking the biodiversity output, the timber production and the conservation measures. Indeed, this relationship represents the production set of multifunctional forests, composed of biodiversity conservation and marketed goods (i.e., timber). The biodiversity output variable BI is thus endogenous in the cost function. Indeed, its level can be explained by several factors such as contractor’s skills (which can be influenced by contractor’s type), the size of enrolled forest area and local land pressure context - which are explanatory variables in the contract cost function - but also by other unobserved factors that cannot

be included in the cost function. Variable BI is thus modeled as a function of timber production, referred to as Y and the set of conservation measures X and other independent variables also affecting conservation costs. All variables are described in Table I.2.

We write the ecological relationship between biodiversity and timber production in the simple following form:

$$\ln(BI) = \omega_0 + \omega_1 Y + \omega_2 X + \omega_3 \ln(LV) + \omega_4 \ln(A) + \omega_5 Z + \epsilon_{BI}. \quad (\text{I.6})$$

Finally, we estimate the following system of two simultaneous equations:

$$\begin{cases} \ln(BI) = g(Y, X, LV, A, Z, \epsilon_{BI}) \\ \ln(C) = f(BI, LV, A, Z, \epsilon_C) \end{cases} \quad (\text{I.7})$$

3.4 Data

In this study, we use data already gathered and used by Tu *et al.* (2013) who collected all relevant information concerning the N2K network in France from several databases. The OSIRIS Database⁸ provides the main information at the contract level: the date on which the contract has been signed, the contract duration⁹, the name and geographic location of the municipality and N2K site where the contract has been signed, contractor type (i.e., private forest owner, municipality, National Forest Agency, etc.), number and type¹⁰ of implemented conservation measures, size of the enrolled forest area, targeted species and habitat and, finally, the total contract's cost being equal to the total payment given to the contractor.¹¹

As explained above (see Section 2.1) we do not observe the use of classical inputs (i.e., capital, labor, energy, material). We instead consider the type of conservation

⁸The OSIRIS database contains precise technical, environmental and socio-economic information about all the 150 N2K forest contracts that have been signed in France between 2007 and 2010, except for personal information about contractors.

⁹Contracts are generally signed for 5 years, except for contracts involving the development of senescent woods, whose duration is 30 years (MEDDE, 2012).

¹⁰Fifteen different types of conservation measures exist and can be implemented in N2K contracts; see Appendix A, Table I.5 for details.

¹¹In a complete information context, the provision costs of biodiversity can be measured by the payment aimed at reimbursing these costs.

measures implemented as well as the number of *hors-barème*¹² conservation measures as determinants of cost variations. Capital invested in biodiversity conservation is proxied by the size of the forest area that is enrolled in the contract and average annual land value per hectare in the municipality where the contract has been implemented. This variable is referred to as *LANDVALUE*. Regarding the variable “size of forest area enrolled in the contract”, we distinguish between areas enrolled for *hors-barème* and *sur-barème* conservation measures.

The Biodiversity Index (*BI*) was built using the Natura 2000 database, which is managed by the French National Inventory of Natural Heritage (INPN) and provides, for each N2K site, an inventory of all species and habitats present in the site at the time of designation, with information about their conservation status.¹³ The French National IUCN Red List of endangered species provides the Red List Status of each species, whereas the Habitats Directive provides information about the Priority Status of habitats and species.

We control for the socio-economic context for each contract in our sample by including socio-economic data in our analysis: average regional timber productivity taken from the 2006 National Forestry Inventory (referred to as *PRODUCTIVITY*), average regional timber harvesting taken from the 2010 national *Agreste* inventory, and average municipal annual land value per hectare taken from the national database of the French Internal Revenue Service (*LANDVALUE*). In order to refine our analysis of the potential effect of timber production on biodiversity provision, we also use the average “regional” timber productivity (*PRODUCTIVITY_SER*) taken from the 2013 National Forestry Inventory calculated for the 91 French *forest ecoregions* (SER)¹⁴.

Finally, we perform our analysis on 140 of the 150 N2K forest contracts signed between 2007 and 2010 in France, and for which we are able to define ecological and socio-economical contexts. The definition of variables and descriptive statistics are presented in Table I.2.¹⁵

¹²In the case of Natura 2000 forest contracts, conservation measures can be implemented in two ways: (1) either in the *sur-barème* mode where payments are defined according to a tariff pay scale; (2) or in the *hors-barème* mode. In the latter case, payment is allocated against the provision of supporting documents (i.e., bills) (MEDDE, 2012).

¹³This information was initially taken from the French Standard Data Forms, which correspond to N2K site “identity cards”, and that were added to the N2K database over a second phase.

¹⁴*Forest ecoregions* have been initially defined by the French National Forestry Inventory in 2011. They correspond to sylvicultural regions that are considered homogeneous in terms of potentiality and conditions for timber production (i.e., climate, soil, presence and share of forest habitats).

¹⁵Only the types of conservation measure that have been implemented in at least 10% of the contracts in our sample were included in our analysis. See Table I.5 for details.

Table I.2: Definition of variables and descriptive statistics

Variable	Definition	Mean	Std. Dev	Min	Max
ACTION F1 ^a	Dummy if ACTION F1 implemented	0.19	0.40	0	1
ACTION F2	Dummy if ACTION F2 implemented	0.14	0.34	0	1
ACTION F3 ^b	Dummy if ACTION F3 implemented	0.09	0.28	0	1
ACTION F5	Dummy if ACTION F5 implemented	0.21	0.41	0	1
ACTION F6	Dummy if ACTION F6 implemented	0.11	0.32	0	1
ACTION F8 ^b	Dummy if ACTION F8 implemented	0.04	0.20	0	1
ACTION F9 ^b	Dummy if ACTION F9 implemented	0.03	0.17	0	1
ACTION F10 ^b	Dummy if ACTION F10 implemented	0.01	0.08	0	1
ACTION F11	Dummy if ACTION F11 implemented	0.18	0.38	0	1
ACTION F12 ^c	Dummy if ACTION F12 implemented	0.39	0.49	0	1
ACTION F13 ^b	Dummy if ACTION F13 implemented	0.02	0.15	0	1
ACTION F14	Dummy if ACTION F14 implemented	0.34	0.48	0	1
ACTION F15 ^b	Dummy if ACTION F15 implemented	0.07	0.26	0	1
COMMUNE	Dummy if the contractor is a municipality	0.27	0.45	0	1
ONF	Dummy if the contractor is a local branch of the NFA ^d	0.28	0.45	0	1
EPCI	Dummy if the contractor is an association of municipalities	0.06	0.25	0	1
SYNDIC_MIXTE	Dummy if the contractor is an association of public actors	0.04	0.20	0	1
PNR ^b	Dummy if the contractor is a regional natural parc	0.01	0.12	0	1
ETS_PUBLIC ^b	Dummy if the contractor is a public institution	0.01	0.08	0	1
PRIVATE_FIRM ^b	Dummy if the contractor is a private firm	0.03	0.17	0	1
PRIVATE_OWNER ^c	Dummy if the contractor is an individual private forest owner	0.18	0.38	0	1
FOREST_GROUP ^b	Dummy if the contractor is a group of private forest owners	0.04	0.20	0	1
FOREST_COOP ^b	Dummy if the contractor is a forest cooperative	0.01	0.08	0	1
ASSOCIATION ^b	Dummy if the contractor is an association	0.06	0.25	0	1
NB_ACTIONS_HB	Number of <i>hors barème</i> implemented conservation measures	1.40	0.91	0.10	6.00
<i>C</i>	Total cost of the contract in euros	19,329.16	18,832.74	662.00	88,507.00
AREA_HB	Size of <i>hors barème</i> enrolled forest area (in ha)	9.46	36.53	0.01	343.00
AREA_SB	Size of <i>sur barème</i> enrolled forest area (in ha)	0.77	3.54	0.01	29.00
PRODUCTIVITY	Average regional timber productivity (m^3 /ha/year)	6.70	2.42	2.29	10.65
PRODUCTIVITY_SER	Average SER-regional timber productivity (m^3 /ha/year)	5.86	2.79	0.10	10.20
<i>BI</i>	Biodiversity value conserved by the contract	0.10	0.30	0.003	2.06
Rarity	Rarity targeted by the contract	0.09	0.17	0.004	1.04
Vulnerability	Vulnerability targeted by the contract	8.42	7.25	1.5	65.25
Species Richness	Species richness targeted by the contract	2.61	2.27	1	12
LANDVALUE	Annual average land value in the municipality in which the contract has been signed	1,671.85	4,558.49	166.76	51,041.1

(N = 140)

^a Dummy variables are specified as 1 if the stated condition is met for the contract.^b Conservation measures' types implemented in less than 10% of the contracts in our sample have not been included in the models as well as contractor's types dimly represented (e.g. ACTION F3, or contractor's type PNR).^c Variables PRIVATE_OWNER and ACTION F12 have been chosen as references.^d National Forest Agency.

4 Results

4.1 Preliminary analysis

While Tu *et al.* (2013) observe a low participation rate from private forest owners, we could observe an unbalanced participation: the majority of contractors (68%) are public agents (the most common types of contractor are municipalities, associations of municipalities and local offices of the National Forest Agency), while two-thirds of the French forest and 54% of the forests in N2K sites are privately owned.

Moreover, N2K forest contracts are implemented in an isolated way (see Figure I.1). Among all forest N2K sites where forest contracts have been implemented, 67% of them reveal only one implemented contract, whereas 32% of them reveal between two and nine implemented contracts. Half of the N2K forest contracts have been implemented in low-biodiversity-value sites¹⁶, whereas some high-biodiversity-value sites with major conservation needs show no implemented N2K forest contracts (see Figure I.1). This shows the low quality of conservation benefits targeting.

Figure I.2 (a) and (b) compare N2K forest contracts's location with average regional timber productivity. A large part of the French N2K forest contracts have been implemented in locations with low timber productivity (i.e., Mediterranean forests, high mountainous areas), and consequently low opportunity costs. Observing opportunity costs related to average municipal land value per hectare gives the same results: N2K forest contracts implementation results from an good cost-targeting. Nevertheless, a few contracts have been implemented in high timber productivity areas such as the *Landes* region. Most of the latter have been signed by public contractors.

Finally, we observe contracts make wise use of the variety of conservation measures. Nevertheless, two types of measures are more frequently implemented than others. Conservation measures F22712 and F22714 (referred to as *ACTION_F12* and *ACTION_F14*, respectively) are implemented in 39% and 34% of observed contracts, respectively (see Appendix A, Table A1).

ACTION_F12 corresponds to the development of senescent wood, which means not harvesting mature trees. *ACTION_F12* is the only type of conservation measure, for which

¹⁶We consider N2K sites as low-biodiversity-value sites when their biodiversity index value is lower than the average biodiversity index value.

allocated payments correspond to the reimbursement of foregone profits in terms of timber production and fully considers opportunity cost. Its abundant implementation could correspond to the reaction of forest landholders to an “appropriate financial compensation” (Boon *et al.*, 2010). Besides, such setting-aside measures are in coherence with forest owners’ values: a great majority of European forest owners importantly value biodiversity aspects of their forests (Boon *et al.*, 2004, 2010) and some of them are willing to set aside forest areas even without any financial compensation (Prokofieva and Gorriz, 2013).

ACTION_F14 corresponds to the establishment of informative signs in order to prevent damaging behaviors and impacts from forest users. The latter is easy to implement. ACTION_F14’s abundant implementation could result from a windfall effect.

4.2 Estimation results and interpretation

We estimate the system (I.7) using Three Stage Least Squares (3SLS) and Full Information Maximum Likelihood (FIML) estimation methods. This makes it possible to handle endogeneity of the biodiversity output variable as well as potential correlation of error terms ϵ_C and ϵ_{BI} . Estimation results obtained with both estimation methods are very similar. The 3SLS method nevertheless provides more efficient estimates by accounting for heteroscedasticity.

In our econometric analysis, we consider several qualitative variables allowing to define categories (i.e., contractor’s type: municipality, private forest owner, etc.; conservation measure’s type: ACTION_F1, ACTION_F2, etc.). In order to determine the impact of these qualitative variables on dependent variables, we performed Wald tests on each “variable block”, for both first-order terms and cross variables. Table I.3 provides details and results of Wald Tests.¹⁷ We re-estimate the system after having excluded variable blocks that show no significant impact on dependent variables.¹⁸

¹⁷Performing Wald tests allows us to test for the existence of a significant effect of qualitative multinomial variables (disaggregated in dummies) on the dependent variable, whereas Student tests allow us to test for the existence of a significantly different effect of individual dummies (compared to the reference).

¹⁸Estimated coefficients related to dummy variables cannot be directly interpreted as the relative effect of the latter on dependent variables (Halvorsen and Palmquist, 1980). Relative effects of dummy variables are calculated based on estimated coefficients using the formula developed by Halvorsen and Palmquist (1980) (see Appendix B for details).

Figure I.1: Targeting biodiversity conservation benefits - Location of N2K forest contracts in France.

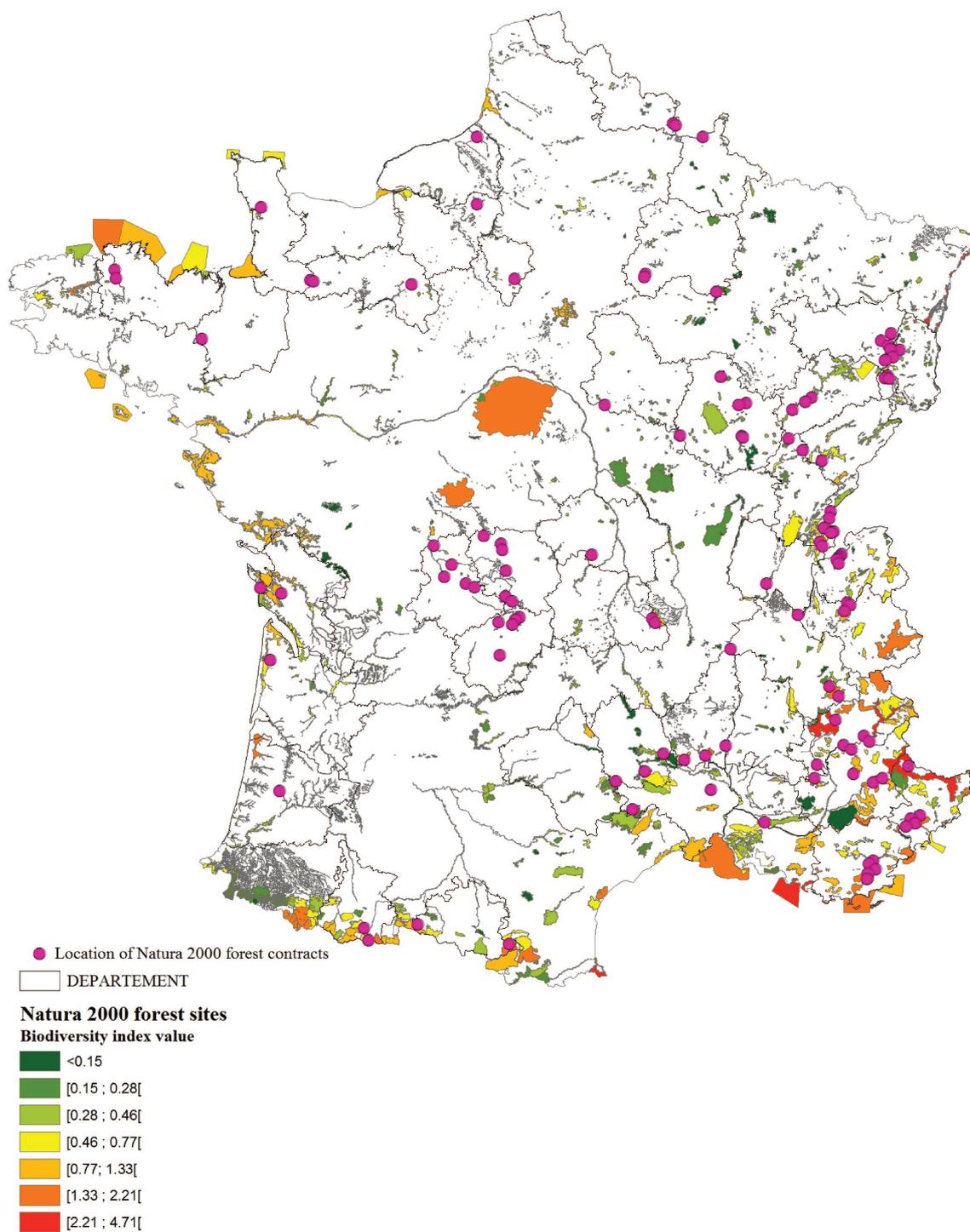
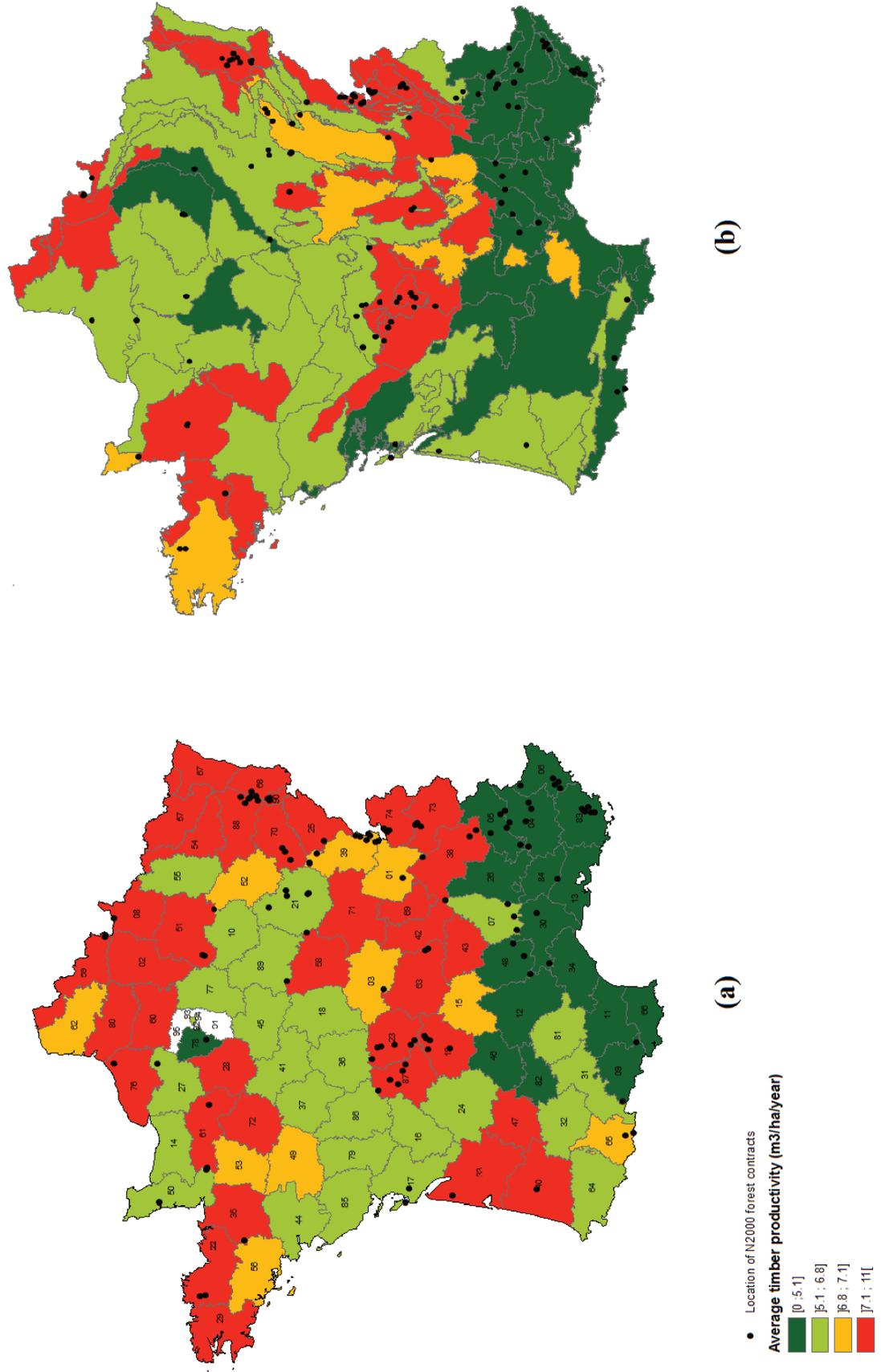


Figure I.2: Targeting opportunity costs - Location of N2K forest contracts in France.



Notes:(a) Average regional timber productivity. (b) Average SER-regional timber productivity.

We also tested for the nullity of coefficients associated with *PRODUCTIVITY* and $BI \times PRODUCTIVITY$ variables. Results show that *PRODUCTIVITY* has no significant influence either on contract's cost, or on cost-elasticity of biodiversity conservation (see Table I.3).

Table I.3 provides two important results: first, the type of conservation measures being implemented in a contract does not have any impact on the latter's total cost. This result could be counterintuitive. However, as shown by the first result of Table I.3, the type of implemented conservation measures has a significant influence on the level of biodiversity output of the contract; thus, including the biodiversity index as the output variable in the cost-function allows us to indirectly capture the effect of conservation measures' type on contract's cost. Second, showing the nullity of coefficients associated to *PRODUCTIVITY* and $BI \times PRODUCTIVITY$ variables allows us to demonstrate that, here, the cost-function for biodiversity provision is separable with respect to timber production. Moreover, this result shows that opportunity cost of timber production is not taken into account in the definition of payments for N2K forest contracts.

Finally, we perform a sensitivity analysis in order to test our results for robustness to the change in the biodiversity index. We then re-estimate our bio-econometric model and take (1) species richness, (2) rarity \times species richness and (3) vulnerability \times species richness as biodiversity output.¹⁹

3SLS estimates are displayed in Table I.4. Values of R^2 for equations *BI* and *C* are 0.53 and 0.61, respectively, thus showing that our econometric model is well adjusted.

Biodiversity equation

Equation (I.6) represents the production set of biodiversity and timber, depending on conservation measures implemented for biodiversity conservation. In equation (I.6), the variable *PRODUCTIVITY* (average regional timber productivity) is a proxy of the timber output. Inputs for biodiversity provision are also considered in equation (I.6) (e.g., type of conservation measures, size of enrolled forest area, average municipal land value, see Section 2.3).

¹⁹Since we have to consider rarity and vulnerability of *all* species/habitats that are targeted by the contract, in order to quantify biodiversity output, species richness is automatically combined with rarity and vulnerability criteria.

Table I.3: Wald test results

Null Hypothesis	Test Statistic (χ^2)	P-value	Decision
Equation (I.6): Biodiversity Index (BI)			
No link between the type of conservation measure and biodiversity output : $\omega_{3ACTIONFi} = 0, \quad \forall i \in \{1; 2; 5; 6; 11; 14\}$	52.27	0.0000	H_0 rejected
No link between contractor's type and biodiversity output : $\omega_{3k} = 0, \quad \forall k \in \{\text{COMMUNE, SYNDIC_MIXTE, ONF, EPCI}\}$	10.41	0.0340	H_0 rejected
Equation (I.5): Contract cost (C)			
No influence of contractor's type $\beta_k = 0, \quad \forall k \in \{\text{COMMUNE, SYNDIC_MIXTE, ONF, EPCI}\}$	13.54	0.0089	H_0 rejected
No influence of contractor's type on cost-elasticity for biodiversity production: $\gamma_{BIk} = 0, \quad \forall k \in \{\text{COMMUNE, SYNDIC_MIXTE, ONF, EPCI}\}$	3.67	0.4529	H_0 accepted
No influence of the type of conservation measure $\beta_{ACTIONFi} = 0, \quad \forall i \in \{1; 2; 5; 6; 11; 14\}$	8.16	0.2265	H_0 accepted
No influence of the type of conservation measure on cost-elasticity for biodiversity production $\gamma_{BIACTIONFi} = 0, \quad \forall i \in \{1; 2; 5; 6; 11; 14\}$	4.20	0.6491	H_0 accepted
No influence of contractor's type on the marginal effect of land value $\delta_{LVk} = 0, \quad \forall k \in \{\text{COMMUNE, SYNDIC_MIXTE, ONF, EPCI}\}$	11.84	0.0186	H_0 rejected
No influence of the type of conservation measure on the marginal effect of land value $\delta_{LVACTIONFi} = 0, \quad \forall i \in \{1; 2; 5; 6; 11; 14\}$	10.61	0.1012	H_0 accepted
No influence of timber productivity: $\alpha_{PRODUCTIVITY} = 0$	0.08	0.7785	H_0 accepted
No influence of timber productivity on cost-elasticity for biodiversity conservation : $\gamma_{BIPRODUCTIVITY} = 0$	0.00	0.9945	H_0 accepted

Table I.4: Estimation results

Variable	Coefficient											
	Equation (5) : Biodiversity conservation											
	BI			Vulnerability			Rarity		Species richness			
ln(PRODUCTIVITY)	-1.083	***	(0.207)	-0.220		(0.137)	-0.990	***	(0.214)	-0.292	**	(0.131)
ACTION_F1	-0.064		(0.216)	0.034		(0.144)	-0.105		(0.224)	-0.046		(0.130)
ACTION_F2	-0.094		(0.225)	0.146		(0.150)	-0.250		(0.233)	0.273	*	(0.139)
ACTION_F5	0.688	***	(0.205)	0.393	***	(0.139)	0.577	***	(0.213)	0.351	***	(0.126)
ACTION_F6	-1.076	***	(0.258)	0.155		(0.171)	-0.829	***	(0.267)	-0.265	*	(0.151)
ACTION_F11	0.987	***	(0.234)	0.156		(0.155)	0.930	***	(0.243)	-0.115		(0.137)
ACTION_F14	-0.567	***	(0.213)	0.042		(0.142)	-0.382	*	(0.221)	-0.041		(0.127)
COMMUNE	0.402	*	(0.209)	-0.094		(0.141)	0.530	**	(0.217)	-0.064		(0.144)
ONF	0.057		(0.206)	0.031		(0.139)	0.282		(0.214)	-0.142		(0.143)
SYNDIC_MIXTE	-0.811	*	(0.427)	-0.007		(0.288)	0.236		(0.443)	-0.759	**	(0.294)
EPCI	0.076		(0.344)	-0.139		(0.231)	-0.127		(0.357)	-0.230		(0.237)
ln(LANDVALUE)	-0.238	**	(0.103)	0.026		(0.070)	-0.113		(0.107)	-0.036		(0.071)
ln(AREA_SB)	0.180	***	(0.045)	0.076	**	(0.031)	0.112	**	(0.047)	0.144	***	(0.031)
ln(AREA_HB)	-2.10 ⁻⁴		(0.033)	0.024		(0.022)	-0.004		(0.035)	0.029		(0.022)
Intercept	-0.076		(0.186)	-0.144		(0.125)	-0.234		(0.193)	0.025		(0.122)
R ²	0.53			0.16			0.41		0.25			
Equation 2 : Contract Cost (C)												
	BI			Vulnerability			Rarity		Species richness			
ln(Biodiversity_variable)	0.350	***	(0.101)	0.559		(0.382)	0.407	***	(0.124)	0.807		(0.743)
ln(Biodiversity_variable) ²	-0.037		(0.058)	-0.011		(0.317)	-0.127		(0.084)	0.260		(1.006)
ln(LANDVALUE)	0.511	***	(0.133)	0.461	***	(0.147)	0.431	***	(0.137)	0.504	***	(0.166)
ln(LANDVALUE) ²	-0.296	***	(0.102)	-0.194	*	(0.106)	-0.237	**	(0.096)	-0.216		(0.161)
COMMUNE	0.215		(0.174)	0.398	**	(0.181)	0.167		(0.176)	0.406		(0.256)
ONF	0.605	***	(0.175)	0.725	***	(0.180)	0.532	***	0.179	0.812	***	(0.237)
SYNDIC_MIXTE	0.738		(0.483)	0.638		(0.523)	0.578		(0.471)	1.259	**	(0.620)
EPCI	0.569	*	(0.313)	0.967	***	(0.320)	0.638	**	(0.304)	1.009	**	(0.456)
ln(LANDVALUE) × COMMUNE	-0.366	*	(0.206)	-0.503	**	(0.221)	-0.367	*	(0.203)	-0.531	*	(0.274)
ln(LANDVALUE) × ONF	-0.445	*	(0.256)	-0.516	*	(0.269)	-0.408		(0.255)	-0.606	*	(0.317)
ln(LANDVALUE) × EPCI	-0.997	**	(0.501)	-0.923	*	(0.534)	-0.840	*	(0.495)	-0.906		(0.631)
ln(LANDVALUE) × SYNDIC	-0.713		(0.871)	-1.409		(0.927)	-1.223		(0.814)	-1.203		(1.214)
ln(AREA_SB)	-0.443	***	(0.156)	-0.441	***	(0.167)	-0.409	***	(0.152)	-0.459		(0.288)
ln(AREA_SB) ²	0.185	***	(0.058)	0.192	***	(0.060)	0.178	***	(0.057)	0.165	*	(0.094)
ln(AREA_HB)	0.122	***	(0.025)	0.134	***	(0.029)	0.128	***	(0.024)	0.114	***	(0.035)
ln(AREA_HB) ²	0.055	***	(0.019)	0.060	***	(0.020)	0.056	***	(0.019)	0.055	**	(0.022)
Intercept	8.596	***	(0.173)	8.411	***	(0.181)	8.69	***	(0.183)	8.362	***	(0.309)
R ²	0.61			0.54			0.62		0.40			

Notes: Standard errors are in brackets.

*, ** and *** stand respectively for 10%, 5% and 1% significance levels.

BI: Biodiversity index value quantifying the level of biodiversity output in terms of targeted biodiversity value *and* conservation needs.

Vulnerability: sum of targeted habitats' and species' vulnerability, quantifying the level of biodiversity output in terms of conservation needs.

Rarity: sum of targeted habitats' and species' rarity. This index quantifies the level of biodiversity output in terms of biodiversity value.

Estimation results of biodiversity equations for several biodiversity indices are presented in the upper part of Table I.4. Values of R-squared are found very different depending on the selected index and show the best fit of data with the biodiversity index *BI* (with a value of 0.53). The rarity index also shows a fair adjustment with a R-squared equal to 0.41.

The coefficient associated with the variable PRODUCTIVITY is significantly different from zero at the 1% level (with a value of -1.083).²⁰ This result is robust whatever the biodiversity index used. This means that a contract implemented in a region with higher average timber productivity allows lower biodiversity conservation benefits. It shows a substitutability relationship between biodiversity conservation and timber production. This can be explained by the fact that forest management aimed at timber production triggers the establishment of particular environmental conditions (i.e., disturbance regime, luminosity, etc.) that favour the development of “ordinary”, resistant and polyvalent species. Such a substitutability phenomenon has already been shown by Armsworth *et al.* (2012) for agri-environment schemes and Rosenkranz *et al.* (2014) within the N2K framework.

LANDVALUE has a significant negative impact on the level of biodiversity output, meaning that lower biodiversity levels are conserved on lands with higher monetary values. Indeed, a higher land value is often associated with higher land pressure as well as a higher urbanization level, which has been identified as a source of native biodiversity loss and biodiversity homogenization (Mckinney, 2002; Mckinney, 2006; Devictor *et al.*, 2007).

Inasmuch our biodiversity index is based on an *ex ante* approach we cannot determine the impact of conservation measures on the level of biodiversity output from an *ex post* perspective. However, the significant positive effect of conservation measures variables on the biodiversity output variable allows us to assess – *ex post* - the coherent targeting and use of each type of conservation measure.

Fifteen different types of conservation measures exist and can be implemented in N2K contracts. However, at the N2K site level, a subset of these 15 measures, which are eligible for forest contracts in this specific site, is defined based on site-specific pre-

²⁰When considering average SER regional timber productivity data, we found that substitutability between timber production and biodiversity conservation is less severe (with a value of -0.783).

existing level of biodiversity and conservation status. The latter are listed in the site's management plan. At the individual contract level, forest owners agree upon the implementation of specific conservation measures they choose within this menu of eligible conservation measures and the link between the type of implemented conservation measures in a contract and targeted species and habitats is made *ex post* by the conservation agency. Choosing to implement a specific conservation measure has an *ex ante* influence of the level of biodiversity output: among eligible conservation measures specified by the management plan, each of them benefits to the conservation of only a part of existing biodiversity in the N2K site.

There exists an official description classifying each of the 15 measure types as a *restoration* or a *conservation* measure (see Table I.5 and MEDDE, 2012). We think that conservation measure F22712, aiming at maintaining and increasing the number of old trees, is susceptible to provide higher conservation benefits than classical conservation measures. This is why we have chosen this measure (referred to as ACTION_F12) as a reference to distinguish between conservation and restoration measures and thus assess the coherent use of the latter: restoration measures should be applied to areas showing higher conservation needs and conservation measures should target lower ones.

Before commenting estimation results, it must be recalled that higher values of *BI* can be linked either to a higher level of the conserved biodiversity value or to a higher level of addressed conservation needs. The different biodiversity indexes used in our analysis make it possible to distinguish between these two effects (see Table I.4): level of conservation needs is measured by Vulnerability, and biodiversity value by rarity and species richness.

ACTION_F1, 2, 5, 6 and 11 are all defined as restoration measures (see Table A.1), thus the latter are supposed to be applied to higher values of *BI* and first to higher level of conservation needs (i.e. higher values of Vulnerability) compared to the reference. Estimation results show the coherent implementation of ACTION_F5, which has been applied to both higher conservation needs and higher biodiversity value. Results show that all considered restoration measures have been applied to higher or similar values of the vulnerability index compared to the reference. It can be noticed that ACTION_F11 has also been applied to higher values of biodiversity value (i.e. Rarity), whereas ACTION_F6 and ACTION_F14 have been applied to lower levels of biodiversity

value compared to the reference. Finally, our results are coherent according to official descriptions of all considered measures and show their coherent use within the N2K forest contract framework, underlining the quality of site-specific management plans.

The identification of contractor's types makes it possible to determine potential differences in targeting a given level of biodiversity conservation. We chose private forest owners as a reference. The significant positive effect of *COMMUNE* shows that municipalities target higher levels of biodiversity conservation than private forest owners. This could be explained by the fact that municipalities are responsible for both the conservation of high biodiversity value sites and for the provision of recreation goods, including biodiversity. In addition, the negative sign of the coefficient of *SYNDIC* indicates that contracts that have been signed by associations of public actors target lower levels of biodiversity benefits than private owners.

Finally, we find a significant positive sign for the size of the forest area, but only for lands enrolled in *sur-barème* conservation measures. Contracts for the conservation of highly vulnerable species and habitats seem to be rather implemented under the *sur-barème* than under the *hors-barème* one. This shows that the conservation of vulnerable species and habitats is organized beforehand through *sur-barème* measures and highlights the quality of management plans.

Cost function for biodiversity conservation

The lower part of Table I.4 presents estimation results of the cost function for different biodiversity indices used as output. The higher values of R-squared are found for the *BI* and vulnerability indices (0.61 and 0.62, respectively).

As previously mentioned (see section 3.2, Table I.3), we show that the *PRODUCTIVITY* variable has no significant influence either on contract cost, or on cost-elasticity for biodiversity conservation. This result confirms the fact that opportunity costs of timber production are not taken into account in the definition of payments for N2K forest contracts, and this, in spite of our previous result on substitutability in production between biodiversity and timber production (see subsection 3.2.1 and Table I.4).

In the translog cost function estimation, coefficients associated with variables in log can be directly interpreted as cost elasticities at the individual mean of variables. The

coefficient associated with the first-order term of the biodiversity index, which represents the cost elasticity of output, is ranged between zero and one. This result is robust whatever the biodiversity index, even though not always significant. We then test the null hypothesis of increasing returns to scale (from the measure of the inverse of cost elasticity). The value of returns to scale for the *BI* index is equal to 2.86 and is shown to be significantly greater than 1 at the 5% level. This suggests that more ambitious contracts - in terms of biodiversity benefits (i.e., conserving a higher biodiversity value) - could be implemented with lower average costs. These results remain valid when considering rarity index as biodiversity output, but do not hold for the last two criteria (i.e., vulnerability and species richness).

Estimations results show positive coefficients indicating higher contract's costs for public contractors (i.e., associations of municipalities, ONF) when conserving a mean level of biodiversity, with respect to private owners. This could be explained by the fact that public contractors are often characterized by higher transactions costs (i.e., administrative costs, learning costs). However, this result could also be explained by unobserved characteristics of the forestlands or landowner characteristics.

Average municipal land value also has a significant positive impact on contract's cost. This result could indicate that land value (that is correlated with opportunity costs of land use) is taken into account in the definition of N2K payment. This result is supported by the ministerial memorandum DGPAAT/SDDRC/C2012-3047 (MEDDE, 2012), that requires that payments for N2K contracts be adjusted to the regional economic context (e.g., land pressure). Moreover, the influence of public contractors on the marginal effect of land value should be uppermost noticed. Cross variables between public contractors and average municipal land value have a significant negative impact on contract's cost. This seems to show that public contractors are willing to bear higher opportunity costs of land use than private contractors.

We investigated the potential impact of two *a priori* important determinants of contract's cost: contracts' duration and type of implemented conservation measures. Our tests concluded to no significant influence of duration on the contract's cost. Furthermore, as previously mentioned (see section 3.2 and Table L.3), results of Wald tests show the non-significant influence of the type of implemented conservation measures on the

total contract's cost. However, including the biodiversity index as the output variable in the cost-function allows us to indirectly account for the impact of conservation measures' type on contract's cost, since the type of implemented conservation measures has a strong influence on the level of biodiversity output of the contract. Moreover, indirectly accounting for the type of implemented conservation measures allows to control for contract's duration since contracts are signed for five years, excepted for ACTION_F12 for which the contract's duration is 30 years.

Finally, the size of the forest area enrolled in the N2K forest contract for *hors-barème* conservation measures has a significant positive effect on the contract cost at the 1% level, whereas the size of the forest area enrolled for *sur-barème* conservation measures has a significant negative impact. The latter are defined beforehand in the management plan, whereas *hors-barème* conservation measures are defined on a case-by-case basis for each contract. Consequently, they are likely to trigger higher costs than *sur-barème* conservation measures (due to, e.g., administrative costs).

5 Discussion and conclusion

Our work allowed us to globally assess the coherence of the implementation of the EU N2K policy in forests in France in the form of contracts and to define potential implications in terms of cost-effectiveness.

In theory, several elements of N2K forest contracts' design are likely to favor cost-effectiveness of the policy (i.e., prioritization rules, flexible payments, monitoring system; see MEDDE (2012)). Basing payment's definition on supporting bills, i.e., in the form of a cost-plus contract (Laffont and Tirole, 1993) allows avoiding informational rents, but is less susceptible to be efficient in terms of cost savings.

At the national scale, we could observe a good quality of conservation costs' targeting. However, conservation benefits have been poorly targeted and participation in N2K contracts is low, which is susceptible to undermine cost-effectiveness of the scheme. Nevertheless, we can conclude on the coherence and good use of N2K sites' individual management plans, which is likely to favor cost-effectiveness. The latter defines - according to site-specific environmental conditions and conservation needs - which of the 15 measures eligible for N2K forest contracts can be implemented within the considered

site. Management plans are also responsible for the definition of *sur-barème* conservation measures. Actually, our results show the coherent implementation of most of the conservation measures' types as well as the interest of *sur-barème* measures for the conservation of highly vulnerable species and habitats.

Simultaneously estimating a cost function for biodiversity conservation as well as a production set for biodiversity conservation and timber allows us to discuss the coherence of the definition of the payment for N2K forest contracts. Results show that potential income losses from timber production are not taken into account in the payment's definition despite strong substitutability in production between biodiversity conservation and timber. This confirms the inaccurate definition of payments for N2K contract, which had already been highlighted by Anthon *et al.* (2010) and Tu *et al.* (2013). The inaccuracy of payments' definition is highlighted by the low participation rate in N2K forest contracts as well as by the abundant implementation of the ACTION_F12: although - as shown by Prokofieva and Gorriz (2013) and Boon *et al.* (2004, 2010) - some private forest owners derive private benefits from senescent areas in their forest and are consequently willing to set-aside forest areas without any financial compensation, the definition of an appropriate financial compensation is decisive in increasing private forest owners' willingness to set-aside productive forest area for conservation (Boon *et al.*, 2010). By failing to trigger participation, payments for N2K forest contracts lead to social inefficiency (Engel *et al.*, 2008). This insufficient definition of payment could lower cost-effectiveness of the N2K forest contracts policy. By leading to low participation it impedes the use of prioritization and targeting rules.

Finally, estimating a cost function allows us to define factors influencing the costs of biodiversity conservation and thus to provide policy recommendations in terms of contracts' targeting. Results show that triggering higher participation from private agents and targeting the latter could be crucial since they show interesting properties in terms of cost-effectiveness for biodiversity conservation. Such results put in perspective with the one obtained by Vedel *et al.* (2015) could lead to question the additionality of N2K forest contracts. Indeed, Vedel *et al.* (2015) show that forest owners' WTA for contracts for ecosystem services provision is sensitive to additionality: forest owners already implementing prescribed management practices aimed at ES provision before

signing a contract have a significantly lower WTA. Then, do N2K contracts consist in paying forest owners to apply management practices they are already used to implement and from which they gain utility? It seems not to be the case: such management practices favouring ES provision are very likely to be implemented by some public forest managers as well, insasmuch they are responsible for the protection of high biodiversity value sites and the provision of recreation goods independently from N2K contracts. This should also lead to lower contract's cost when following Vedel *et al.* and no differences in terms of cost-effectiveness should be observed. Thus, our results do point better cost-effectiveness performances of private owners for biodiversity provision. Moreover, targeting contracts towards public landholders in high land-pressure areas could enhance cost-effectiveness of the policy since we show that the latter are willing to bear higher opportunity costs linked to land value.

Results show the opportunity to exploit some economies of scale related to biodiversity conservation, thus prioritizing contracts that are likely to provide a higher level of biodiversity value (i.e., species richness and rarity) would also enhance cost-effectiveness (see Section 3.3.2).

In view of these results, redefining payments seems to be necessary. The latter should be in coherence with a multifunctional vision of the forest by taking positive and negative interactions between timber production and biodiversity conservation into account, as it is already done for payments for agri-environment measures. Finally, integrating opportunity costs linked to timber production in the definition of payments for N2K forest contracts should be made by taking "SER-regional" regional timber productivity data into account. This new payment definition could enhance the policy's cost-effectiveness. However, this would also trigger additional transaction costs that should be compared to cost-effectiveness gains.

Our study reveals some limits that should be addressed in future research. First, our biodiversity index is based on information taken from the Standard Data Forms (see Section 3.3). These data are homogeneous, nevertheless their reliability has been questioned (Marage, 2013). In the future, we should perform our analysis using data from an initial conservation status assessment that is included in the management plan of each N2K site. This information seems to be more reliable but heterogeneous (Yon *et*

al., 2003; Marage, 2013). Finally, an adequate cost-effectiveness evaluation of the N2K forest contracts schemes should be based on an *ex post* approach, allowing to directly measure the ecological impact of conservation measures on the conservation status of targeted species and habitats.

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7 Appendix

7.1 Eligible conservation measures for Natura 2000 forest contracts

Table I.5: Eligible conservation measures for Natura 2000 forest contracts

Official Code	Variable name	Official description	Restoration/ Conservation	Implementation rate
F22701	<i>ACTION_F1</i>	Creation or re-establishment of clearings and moorlands	Restoration	19.29%
F22702	<i>ACTION_F2</i>	Creation or re-establishment of forest ponds	Restoration	13.57%
F22703	<i>ACTION_F3</i>	Implementation of human-managed regeneration	Conservation	8.57%
F22705	<i>ACTION_F5</i>	Marking, felling or pruning operations without timber production objectives	Restoration	21.43%
F22706	<i>ACTION_F6</i>	Maintenance and restoration operations of riparian forests, riverbanks vegetation and reasoned removal of (log)jams - Productive or non-productive context	Restoration	11.43%
F22708	not included	Manual extrication or weeding operations in place of chemical extrication or weeding operations	Conservation	4.29%
F22709	not included	Support for additional costs triggered by investments aiming at lowering forest service roads' impact	Conservation and Restoration	2.83%
F22710	not included	Fencing habitats types of Community interest	Restoration	0.71%
F22711	<i>ACTION_F11</i>	Operations aiming at eliminating or containing invasive species	Restoration	17.86%
F22712	<i>ACTION_F12</i>	Favoring the development of senescent (large) trees (single trees or small areas)	Reference	38.57%
F22713	not included	Innovative operations in favor of species and habitats	Restoration	2.14%
F22714	<i>ACTION_F14</i>	Investments aiming at informing forest users	Conservation	34.29%
F22715	<i>ACTION_F15</i>	"De-regularization" operations of forest stands with no timber-production objective	Restoration	7.14%
F22716	not included	Support for additional costs triggered by the implementation of alternative skidding methods	Conservation	0%

From Hesmatol Vaezin *et al.* (2014).

7.2 Relative effects of dummy variables

Estimated coefficients associated with dummy variables cannot be directly interpreted as the relative effect of the latter on dependent variables (Halvorsen and Palmquist, 1980). Relative effects of dummy variables are calculated based on estimated coefficients (here β_k) using the formula developed by Halvorsen and Palmquist (1980).²¹ Let g_k be the relative effect. We have:

$$g_k = \exp(\beta_k) - 1$$

Table I.6: Re-calculated relative effects of dummy variables on the contract cost

Variable	Estimated coefficient β_k	Estimated standard error	Estimated relative effect g_k	Estimated standard error	Percentage effect
ONF	0.605 ***	0.175	0.831 ***	0.321	83%
EPCI	0.569 *	0.313	0.766	0.553	77%
Ln(LANDVALUE) × COMMUNE	-0.366 *	0.206	-0.306 **	0.141	-0.31%
Ln(LANDVALUE) × ONF	-0.445 *	0.256	-0.359 **	0.164	-0.36%
Ln(LANDVALUE) × EPCI	-0.997 *	0.501	-0.631 ***	0.184	-0.63%

Note: *, **, *** stand for 10%, 5% and 1% significance levels, respectively.

²¹Standard errors are computed using the delta method.

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Chapter II

Designing species-specific conservation contracts in a heterogeneous landscape with unobservable conservation costs and benefits

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Abstract

Paying for species-specific conservation requires to encourage landowners to provide both habitat suitability in order to establish conservation networks, and species protection in order to maintain its presence. We investigate the possibility to define differentiated contracts for species-specific conservation when both conservation costs and benefits are unobservable and heterogeneous. We develop a *common-value* principal-agent model, in which the principal's preferences for both types of conservation benefits is explicitly taken into account. The level of effective protection benefits provided by an agent is captured by her level of unobservable protection costs. We analytically demonstrate the possibility to define differentiated conservation payments despite a non-responsiveness situation, known to usually lead to bunching equilibria. Results of numerical landscape-scale simulations show that contracts derived from a *common-value* model can perform better than those derived from a classic adverse selection model. We find differentiated contracts, which are closer to first-best ones and show interesting cost-effectiveness performances.

Keywords: biodiversity conservation, adverse selection, principal-agent models, common-value models

1 Introduction

Despite the adoption of the EU biodiversity strategy to 2020 and the almost complete implementation of the Natura 2000 network, the overall status of biodiversity in the European Union (EU) has not significantly improved during the 2008-2012 period as mentioned in the last report on the State of Nature in the EU (EAA, 2015). However, this report highlights the success of targeted species-specific conservation programs such as EU Species Action Plans in delivering substantial conservation results. The effectiveness of species-specific conservation programs had already been underlined by Hoffmann et al. (2010) for vertebrate species. The implementation of targeted species-specific conservation programs by Member States is required under the EU Habitat and Birds directive framework for certain rare and threatened species of Community interest. Such targeted conservation programs are usually based on a twofold conservation objective (EC, 2007).¹ They aim, on the one hand, at the provision of suitable and sufficient habitat area for the species;² on the other hand, they aim at the physical protection of individual specimens as well as the conservation of core breeding and resting sites.

Both guidance documents of the European Commission (EC) and studies from the conservation literature stress the need to involve local stakeholders in the implementation of such species-specific conservation programs and underline the interest of existing payment for ecosystem services (PES) schemes for the establishment of flexible conservation networks (Orbicon et al., 2009). Indeed, existing schemes already contribute to enhance habitat suitability and landscape connectivity inside and outside protected areas by giving landowners economic incentives to improve management practices (Donald and Evans, 2006; Heller and Zavaleta, 2009).

PES schemes for biodiversity conservation target landowners showing heterogeneous preferences, management practices and skills regarding biodiversity conservation. Land properties are also heterogeneous in terms of land suitability and quality regarding

¹Numerous animal and vegetal species are listed in both Annex II and IV of the EU Habitat directive. These species are therefore covered by the habitat protection provisions as well as by the strict provision system. For instance, 572 plant species are listed in Annex II and all of them (except bryophytes) are automatically listed in Annex IV (EC, 2007).

²Each species has specific requirements corresponding to a given set of resources, climate and other ecological conditions. This set of environmental conditions describes the species' ecological niche. On that basis and its environmental characteristics, any geographical location can be characterized in terms of habitat suitability for a target species (Hirzel et al., 2006).

conservation targets. This double heterogeneity of private owners and properties results in heterogeneity of conservation costs and benefits. Moreover, the level of costs and benefits associated with the conservation of a specific property is usually unobservable to the conservation planner and belongs to landowners' private information. These issues have been highlighted as major concerns to be tackled in the definition of efficient PES schemes for biodiversity conservation (Ferraro, 2008, 2011; Hanley et al., 2012).

When implementing species-specific conservation payments, the conservation planner could be legitimately willing to differentiate payments according to the level of conservation costs and benefits and prioritize payments towards landowners providing effective species-specific protection services. Such a differentiation of payments would enhance the cost-effectiveness of the conservation scheme (Armsworth et al., 2012).³ However, the conservation planner could incur a high cost when trying to observe *in situ* the effective presence or absence of target species' populations in private properties. In this context we study the definition of optimal conservation payments for a species-specific conservation program and investigate the possibility to define differentiated conservation contracts while accounting for unobservable and heterogeneous conservation costs and benefits.

Literature applying contract theory to the design of agri-environment schemes (AES) and PES addresses – among other issues – adverse selection problems. In most of these studies, asymmetric information solely concerns conservation costs and the principal's objective is to achieve cost-effectiveness. Agents are screened on a cost-basis (e.g., Moxey et al., 1999; Gren, 2004; Canton et al., 2009). Conservation benefits are usually considered as observable to the principal and are either contractible (e.g., Feng, 2007) or directly stem from the level of contractible effort (e.g., Smith and Shogren, 2002; Gren, 2004; Canton, 2009). Anthon et al. (2010) account for the heterogeneity and the unobserved character of landowners' ability to achieve a high level of conservation benefits for their forest through conservation, but consider conservation benefits as observable and verifiable. Polasky and Doremus (1998) study the impact of various legal and compensation rules on the achievement of efficient land-use decisions (conservation vs. use on production purposes) by a landowner when the species conservation value of

³Cost-effectiveness can be defined as the achievement of the conservation goal at least cost or as the maximization of conservation results for a given budget.

the land parcel is unknown.⁴ Polasky and Doremus (1998) also investigate the impact of asymmetric information on landowner's opportunity cost of conservation. However, the species conservation value of the land parcel is unknown to both the landowner and the regulator. Besides, authors do not investigate the definition of differentiated contracts.

In this chapter, we consider a principal-agent framework, in which a conservation planner (i.e., the principal) delegates the implementation of conservation measures to a landowner (i.e., the agent). The unobservable conservation cost of the landowner is two-dimensional as the sum of opportunity and protection costs. The conservation planner pursues a twofold conservation objective by seeking the provision of both habitat suitability and species-specific protection services. We render principal's twofold conservation objective by developing a *common-value* model,⁵ in which the principals' preference for both types of conservation benefits is explicitly taken into account. The level of species-specific protection service provided by an agent is captured by her level of unobservable protection costs. Hence, principal's utility from contracting increases according to the amount of conserved area and the level of protection cost. We compare conservation contracts derived from a *common-value* model with those obtained from a benchmark model involving classic adverse selection. Finally, we simulate the implementation of biodiversity conservation contracts at the landscape scale and compare the cost-effectiveness of contracts derived from these two types of models.

Our contribution is twofold. First, we account for the unobservable character of conservation benefits as well as multidimensionality of asymmetric information, which has been rarely done in the literature applying contract theory to PES/AES design. Feng (2007) developed a principal-agent model for the design of payments in the framework of an AES with a dual policy goal: the principal seeks conservation and income support for small farms. Both conservation cost and farm size are considered unobservable. However, no interdependence between policy goals is accounted for and principal's gains from contracting are defined through two separate value functions. Here, we account for interdependencies between conservation objectives through a bivariate value function to describe principal's gains from contracting. Finally, our work comes closest to

⁴In this chapter, the species conservation value is unknown and related to the occupation or use of the parcel by a protected species (e.g., as a breeding site). The species conservation value of a land parcel can be determined through monitoring.

⁵See Laffont and Martimort (2002), p. 53.

Morand and Thomas (2006) studying optimal procurement mechanisms. They consider a *common-value* principal-agent model with a single value function and only one instrument despite a two-dimensional asymmetric information. However they consider the purchase of an indivisible good.

Second, we develop an analytical framework, which is more comprehensive and better suited to the definition of optimal contracts for species-specific conservation. As we show, basing the definition of such contracts on a classic adverse selection analytical framework would lead to the definition of inefficient contracts.

The chapter is structured as follows. Section 2 presents model's background and main assumptions. Section 3 presents first best contracts. Section 4 defines optimal contracts under asymmetric information. Section 5 presents the methodology and results of simulations performed at the landscape scale. Section 6 discusses our main results and concludes.

2 The model

2.1 Model background and main assumptions

The two-dimensional conservation cost of the landowner

We consider a risk neutral conservation planner - the principal - planning a species-specific conservation program in a landscape. The conservation planner wishes to contract on the implementation of conservation measures by a landowner on an area A in exchange of a conservation payment, henceforth referred to as a transfer t . The amount of conserved area A is assumed to be observable and verifiable. We have $A < A_{tot}$, with A_{tot} being the total area of the owner's property. The principal plans the conservation of a specific target species and pursues a twofold conservation objective: (1) providing suitable habitat and (2) protecting and maintaining existing populations of the target species. Consequently, the principal delegates the implementation of the following types of conservation measures to the agent: (i) provision of suitable habitat through the implementation of prescribed management practices on the whole area A and (ii) locally implementing species-specific protection measures. When contracting, *both* types of conservation measures have to be implemented by the agent.

The unobservable conservation cost incurred by the agent when contracting is described by the function $C(A, \theta^{os})$, with θ^{os} the marginal conservation cost per ha. Conservation cost increases with the amount of conserved area A as well as with the type θ^{os} , so that we have: $C_A > 0$ and $C_{\theta^{os}} > 0$. We also assume that the marginal conservation cost of a hectare increases with the type index θ^{os} , we have: $C_{A\theta^{os}} > 0$.⁶ We consider a two-dimensional conservation cost θ^{os} as the sum of opportunity cost θ^o and species-specific protection cost θ^s ; we have : $\theta^{os} = \theta^o + \theta^s$.

Providing suitable habitat requires the agent to deviate from his standard management practices and makes the agent bear opportunity costs. We have agent's marginal opportunity cost $\theta^o \in \{\theta_L^o; \theta_H^o\}$, with θ_L^o (θ_H^o respectively) denoting a low (high) level of opportunity cost. Marginal opportunity cost θ^o depends on land characteristics and unobservable individual landowner's characteristics (e.g., individual preferences towards biodiversity conservation and production activities, individual management skills). Let ν be the proportion of agents with a low level of opportunity cost in the landscape. We have : $P(\theta^o = \theta_L^o) = \nu$ and $P(\theta^o = \theta_H^o) = 1 - \nu$. Providing suitable habitat could for instance require a landowner to decrease her use of pesticides or fertilizers or to stop using them.⁷

We consider the species-specific marginal protection cost of the agent $\theta^s \in \{\theta_A^s; \theta_P^s\}$, with θ_A^s denoting a low level of protection cost incurred in the *absence* of species individuals on the agent's property. On the contrary, θ_P^s , denotes a high level of marginal protection cost incurred in the *presence* of species individuals. We consider that the landscape is homogeneous in terms of pedoclimatic conditions. Therefore, we consider that species' fitness is homogeneous over the landscape and assume that the level of protection cost only depends on unobservable owner's characteristics (e.g., environmental management skills) and on the unobservable absence/presence of species individuals in the property. Thus, the level of protection cost is directly related to the realization of a part of conservation benefits.⁸ Let μ be the proportion of properties showing

⁶This assumption corresponds to the *Spence-Mirrlees'* property, which has been commonly applied in this strand of literature (e.g. Feng, 2007; Canton et al., 2009, etc.).

⁷The consultant's report for the EC entitled *How Species conservation can be supported through Rural Development Programmes* report (Orbicon et al., 2009) provides several examples of practices and conservation measures, which are or could be implemented under a species-specific conservation program in the EU for 12 endangered animal species. One of the most cited measure consist in reducing or ban the use pesticides and fertilizers in areas where a species is or could be present.

⁸We hypothesize that the impact of the presence of the species dominates the impact of poor landowner's environmental management skills on the marginal protection cost θ^s . Then, a higher level

the *absence* of species individuals in the landscape. We have : $P(\theta^s = \theta_A^s) = \mu$ and $P(\theta^s = \theta_P^s) = 1 - \mu$; $1 - \mu$ can be interpreted as the probability of presence of the target species in the landscape. In absence of the species on the property, providing species-specific conservation protection services could for instance consist in conducting regular monitoring to detect the arrival of species' individuals. In presence of the species in a property, the implementation of specific mowing and grazing practices as well as the establishment of unharvested buffer zones around nests are common protection measures, which are defined for the protection of bird species (Orbicon et al., 2009).

We assume no correlation between the two cost parameters θ^o and θ^s : they are independently distributed and θ^{os} is separable in the two cost parameters.⁹ Thus, $\theta^{os} \in \{\theta_{LA}^{os}; \theta_{LP}^{os}; \theta_{HA}^{os}; \theta_{HP}^{os}\}$ and an agent can correspond to one of the four following types:

Table II.1: Two types of cost - Four types of agents

Agent's type	Cost parameter	Opportunity cost	Species' presence/absence	Probability
LA	θ_{LA}^{os}	Low	Absence	$\nu\mu$
LP	θ_{LP}^{os}	Low	Presence	$\nu(1 - \mu)$
HA	θ_{HA}^{os}	High	Absence	$(1 - \nu)\mu$
HP	θ_{HP}^{os}	High	Presence	$(1 - \nu)(1 - \mu)$

Since we consider that θ^{os} is separable in θ^o and θ^s , the scheduling of conservation costs is not straightforward. We denote by $\Delta\theta^s = \theta_P^s - \theta_A^s \geq 0$ and $\Delta\theta^o = \theta_H^o - \theta_L^o \geq 0$ the difference between high and low levels of protection and opportunity cost respectively. Under the assumption $\Delta\theta^s < \Delta\theta^o$ as well as $C_{A\theta^{os}} > 0$, conservation costs θ^{os} are ordered as follows:¹⁰

$$\theta_{LA}^{os} < \theta_{LP}^{os} < \theta_{HA}^{os} < \theta_{HP}^{os} \quad (\text{II.1})$$

Under the alternative assumption $\Delta\theta^s > \Delta\theta^o$ and $C_{A\theta^{os}} > 0$, conservation costs θ^{os} are ordered in the following way:

of protection cost (i.e., θ_P^s) does signal the presence of the target species.

⁹ Thus, the *Spence Mirrlees'* property can be extended to the θ^o and θ^s cost parameters, which gives: $C_{A\theta^s} > 0$ and $C_{A\theta^o} > 0$.

¹⁰ Assuming $\Delta\theta^s < \Delta\theta^o$ implies that the increase in opportunity cost of conservation triggered by a high-level of opportunity cost (i.e., $\Delta\theta^o$) exceeds the increase in species-specific protection cost triggered by the presence of the species (i.e., $\Delta\theta^s$).

$$\theta_{LA}^{os} < \theta_{HA}^{os} < \theta_{LP}^{os} < \theta_{HP}^{os} \quad (\text{II.2})$$

Finally, for simplicity reasons, we adopt the following functional form for the cost function: $C(A, \theta^{os}) = A \times \theta^{os}$. Thus, θ^{os} corresponds not only to the marginal conservation cost but also to the average conservation cost per hectare.

The utility function of the conservation planner

The principal derives utility from the level of habitat suitability service provided by each contract, which is observable through the amount of conserved area A . Let V denote the principal's utility derived from conservation contracts. Principal's utility is assumed to be increasing and concave with respect to the amount of conserved area and we have: $V_A \geq 0$, $V_{AA} \leq 0$.

We consider a first type of model, in which the principal's objective is to maximize the amount of area under conservation at least cost. Such a model corresponds to a classic adverse selection model (AS). In this case, the principal's utility depends on the amount of conserved area only and can be described by the following function: $V(A)$.

We compare conservation contracts derived from the AS model - *our benchmark* - to conservation contracts derived from a second type of model, in which the principal pursues a twofold conservation objective. In this case, the principal derives utility not only from the level of habitat suitability service provided by each contract but also from the level of provided species-specific protection service.¹¹ As in the AS model, principal's utility is assumed to be increasing and concave with respect to the amount of conserved area and we have: $V_A \geq 0$, $V_{AA} \leq 0$. The level of unobservable species protection cost θ^s signals the absence/presence of target species' individuals in the property. Here, the principal's utility is directly influenced by the level of protection cost θ^s . We assume that principal's utility increases with respect to the value of protection cost and we have:

Assumption A1: $V_{\theta^s} > 0$

¹¹In the context of a species-specific conservation program, land conservation does not generate any utility *per se* but generates utility through the provision of suitable habitat to the target species and its protection. A decision maker derives utility from conservation programs through this effective protection of target-species individuals, which are granted an economic value by society (e.g., existence value, recreational value, etc.).

Besides, we follow Laffont and Martimort (2002) by assuming that marginal utility derived from a conserved hectare quickly increases with θ^s and we have:

Assumption A2: $V_{A\theta^s} > 1$

This denotes a higher productivity, in terms of conservation, of a conserved hectare with presence of the species. This second type of model corresponds to a *common-value* model (CV) and the principal's utility can be described by the following function: $V(A, \theta^s)$.

3 Optimal contracts under full information

3.1 The adverse selection model (AS)

Under full information the principal is able to observe agent's conservation cost and defines conservation contracts $(A(\theta^{os}), t(\theta^{os}))$ so as to maximize the difference between utility derived from contracting and transfers:

$$V(A(\theta^{os})) - t(\theta^{os}) \quad (\text{II.3})$$

Let $R_{\theta^{os}}(\theta^{os})$ denote the rent of an agent of type θ^{os} choosing a contract of type θ^{os} . This rent corresponds to the difference between the transfer received by an agent and its true conservation cost. The definition of conservation contracts is subject to the following agent's individual rationality constraint (i.e., the agent cannot get a negative utility from contracting):

$$R_{\theta^{os}}(\theta^{os}) = t(\theta^{os}) - A(\theta^{os}) \times (\theta^o + \theta^s) \geq 0 \quad (\text{II.4})$$

We derive first-order optimality conditions giving first-best solutions for the AS model. First-best allocation $A^{AS^*}(\theta^{os})$ (i.e., area to be conserved by each type of agent at the first-best) is defined by the following optimality condition $V_A(A^{AS^*}(\theta^{os})) = \theta^o + \theta^s$. Optimal transfers $t^*(\theta^{os})$ are defined in the following way : $t^*(\theta^{os}) = A^{AS^*}(\theta^{os}) \times (\theta^o + \theta^s)$, meaning that the transfer has to cover the whole conservation cost and $R_{\theta^{os}}(\theta^{os}) = 0$ for all type θ^{os} .

Proposition 1:

Under full information and with an AS model, the principal generates conservation at least cost and first-best allocations are classically ordered according to cost-efficiency. The amount of area to be conserved by an agent decreases with the type index θ^{os} : largest areas are to be conserved by the most cost-efficient agents.¹²

Under the assumption $\Delta\theta^s < \Delta\theta^o$ and $C_{A\theta^{os}} > 0$, first-best contracts are ordered in the following way:¹³

$$A_{LA}^{AS*} > A_{LP}^{AS*} > A_{HA}^{AS*} > A_{HP}^{AS*} \quad (\text{II.5})$$

3.2 The common-value model (CV)

In the CV model and similarly to the AS model, the principal aims at maximizing the difference between utility derived from contracting and transfers:

$$V(A(\theta^{os}), \theta^s) - t(\theta^{os}) \quad (\text{II.6})$$

The definition of conservation contracts under full information for the CV model is subject to the agent's individual rationality constraint previously defined in equation (II.4). First-best allocation $A^{CV*}(\theta^{os})$ (i.e., area to be conserved by each type of agent at the first-best) is defined by the following optimality condition $V_A(A^{CV*}(\theta^{os})) = \theta^o + \theta^s$. Similarly to the AS model, optimal transfers $t^*(\theta^{os})$ are defined in the following way : $t^*(\theta^{os}) = A^{CV*}(\theta^{os}) \times (\theta^o + \theta^s)$. Thus, the principal's program amounts to maximizing a social welfare function $W(A(\theta^{os}), \theta^o, \theta^s)$, with $W(A(\theta^{os}), \theta^o, \theta^s) = V(A(\theta^{os}), \theta^s) - A(\theta^{os}) \times (\theta^o + \theta^s)$.

Differentiating $W_A(A(\theta^{os}), \theta^o, \theta^s)$ at the optimum $A^{CV*}(\theta^{os})$ while considering optimality condition $V_A(A^{CV*}(\theta^{os}), \theta^s) = \theta^o + \theta^s$, assumption A2 and assuming $V_{A\theta^o} = 0$ allows us to derive the following lemmas:

Lemma 1:

$$\frac{dA^{CV^*}(\theta^{os})}{d\theta^s} = \frac{1 - V_{A\theta^s}}{V_{AA}} > 0 \quad (\text{II.7})$$

First-best allocation $A^{CV^}(\theta^{os})$ increases with the value of protection cost θ^s (i.e., with the effective presence of target species individuals on the property).*

Proof: see Appendix 8.1

Lemma 2:

$$\frac{dA^{CV^*}(\theta^{os})}{d\theta^o} = \frac{1}{V_{AA}} < 0 \quad (\text{II.8})$$

First-best allocation $A^(\theta^{os})$ decreases according to opportunity cost of conservation θ^o .*

Proof: see Appendix 8.1

Lemma 3:

Under assumption A2 and considering concavity of $V(A(\theta^{os}), \theta^s)$ according to $A(\theta^{os})$:

$$\frac{dA^{CV^*}(\theta^{os})}{d\theta^s} - \frac{dA^{CV^*}(\theta^{os})}{d\theta^o} = -\frac{V_{A\theta^s}}{V_{AA}} \geq 0 \quad (\text{II.9})$$

Hence, the effect of protection cost θ^s dominates the effect of opportunity cost θ^o for the definition

of first-best allocations.

Proof: see Appendix 8.1

We finally derive the following proposition:

Proposition 2:

Under full information, the presence of target-species individuals on the property is always favored over cost-efficiency for the definition of first-best allocations under the CV model. Conservation contracts have to be prioritized towards properties hosting species individuals and we have:

$$A_{LP}^{CV*} > A_{HP}^{CV*} > A_{LA}^{CV*} > A_{HA}^{CV*} \quad (\text{II.10})$$

Thus, with a CV model *LP*- and *HP*-type landowners are asked to conserve the largest areas, and this irrespective of the relationship assumed between $\Delta\theta^s$ and $\Delta\theta^o$.

4 Optimal contracts under asymmetric information

4.1 The adverse selection model

Program of the conservation planner

Under asymmetric information, the principal has to account for the hidden character of cost parameters θ^o and θ^s and defines conservation contracts $(A(\theta^{os}), t(\theta^{os}))$ so as to maximize the difference between *expected* conservation benefits and *expected* transfers $E(W)$:

$$\begin{aligned} E(W) = & \mu\nu[V(A_{LA}) - t_{LA}] + (1 - \mu)\nu[V(A_{LP}) - t_{LP}] \\ & + \mu(1 - \nu)[V(A_{HA}) - t_{HA}] + (1 - \mu)(1 - \nu)[V(A_{HP}) - t_{HP}] \end{aligned} \quad (\text{II.11})$$

The definition of conservation contracts $(A(\theta^{os}), t(\theta^{os}))$ is subject to the agent's individual rationality constraint (see equation (II.4)) as well as to the agent's incentive constraint. As defined earlier, $R_{\theta^{os}}(\theta^{os})$ denotes the rent of an agent of type θ^{os} choosing a contract of type θ^{os} . Let $R_{\theta^{os}}(\widehat{\theta}^{os}) = t(\widehat{\theta}^{os}) - A(\widehat{\theta}^{os}) \times \theta^{os}$ be the rent of an agent of type θ^{os} choosing a contract of type $\widehat{\theta}^{os}$. A θ^{os} -type landowner is better off when contracting truthfully if and only if:

$$R_{\theta^{os}}(\theta^{os}) = t(\theta^{os}) - A(\theta^{os}) \times \theta^{os} \geq R_{\theta^{os}}(\widehat{\theta}^{os}) = t(\widehat{\theta}^{os}) - A(\widehat{\theta}^{os}) \times \theta^{os} \quad (\text{II.12})$$

Accounting for the agent's incentive constraint (II.12) allows to ensure truthful contracting. Twelve incentive constraints have to be considered (see Appendix 8.2). Adopting the *Spence-Mirrlees'* property allows us to only consider local constraints (i.e., incen-

tive constraints between adjacent types) and call for checking for the global constraints *ex post* (Laffont and Martimort, 2002).¹⁴ Moreover, considering that an agent has the incentive to only lie upwards (i.e., to choose contracts designed for less cost-efficient agents), only R_{LA} , R_{LP} and R_{HA} incentive constraints are binding (see equations hereinafter). These constraints correspond to the incentives for LA -, LP - and HA -type agents to locally lie upwards respectively. Only the individual rationality constraint of the less cost-efficient type (i.e., HP -type) is binding (i.e., $R_{HP} = 0$, with R_{HP} denoting the rent of a HP -type agent).

The following information rents have to be considered. These rents are increasing according to cost-efficiency (i.e., decreasing with the type index θ^{os}). See Appendix 8.2 for the definition of information rents.

$$R_{LA} = A_{LP} \times \Delta\theta^s + A_{HA} \times (\Delta\theta^o - \Delta\theta^s) + A_{HP} \times \Delta\theta^s \quad (\text{II.13a})$$

$$R_{LP} = A_{HA} \times (\Delta\theta^o - \Delta\theta^s) + A_{HP} \times \Delta\theta^s \quad (\text{II.13b})$$

$$R_{HA} = A_{HP} \times \Delta\theta^s \quad (\text{II.13c})$$

$$R_{HP} = 0 \quad (\text{II.13d})$$

Integrating binding incentive and participation constraints (II.13a) to (II.13d) into equation (II.11) allows to rewrite the program of the principal. Under asymmetric information and with an AS model, the principal has to solve the following program:

$$\begin{aligned} \max_{(A(\theta^{os}), R(\theta^{os}))} \quad & EW_{(A(\theta^{os}), R(\theta^{os}))} = \mu\nu[V(A_{LA}, \theta_A^s) - A_{LA} \times (\theta_L^o + \theta_A^s)] \\ & + (1 - \mu)\nu[V(A_{LP}, \theta_P^s) - A_{LP} \times (\theta_L^o + \theta_P^s)] \\ & + \mu(1 - \nu)[V(A_{HA}, \theta_A^s) - A_{HA} \times (\theta_H^o + \theta_A^s)] \\ & + (1 - \mu)(1 - \nu)[V(A_{HP}, \theta_P^s) - A_{HP}(\theta_H^o + \theta_P^s)] \\ & - [\mu\nu \times R_{LA} + (1 - \mu)\nu \times R_{LP} + \mu(1 - \nu) \times R_{HA} + (1 - \mu)(1 - \nu) \times R_{HP}] \end{aligned} \quad (\text{II.14})$$

We define first order conditions, displayed in equations (II.15a) to (II.15d) herein-

¹⁴It can be easily shown that the validity of local incentive constraints is a sufficient condition for the validity of global incentive constraints.

below.

$$V_A(A_{LA}^{SB}) = \theta_L^o + \theta_A^s \quad (\text{II.15a})$$

$$V_A(A_{LP}^{SB}) = \theta_L^o + \theta_P^s + \frac{\mu \times \Delta\theta^s}{1 - \mu} \quad (\text{II.15b})$$

$$V_A(A_{HA}^{SB}) = \theta_H^o + \theta_A^s + \frac{\nu \times (\Delta\theta^o - \Delta\theta^s)}{(1 - \nu)\mu} \quad (\text{II.15c})$$

$$V_A(A_{HP}^{SB}) = \theta_H^o + \theta_P^s + \frac{(\nu + \mu(1 - \nu))\Delta\theta^s}{(1 - \mu)(1 - \nu)} \quad (\text{II.15d})$$

Second-best (SB) conservation contracts $(A^{SB}(\theta^{os}), R(\theta^{os}))$ are defined so that, principal's marginal utility derived from a conserved hectare (i.e., $V_A(A^{SB}(\theta^{os}))$) equals virtual marginal conservation cost. Virtual marginal cost corresponds to the sum of the standard marginal conservation cost θ^{os} and information cost, which corresponds to the last term of the right hand side of equations (II.15a) to (II.15d) (e.g., $\frac{\mu \times \Delta\theta^s}{1 - \mu}$ for a *LP*-type agent).

Conditions for the emergence of separating equilibria

Under an AS model, optimal second-best conservation contracts result in a fully separating equilibrium (i.e., a menu of individually differentiated contracts) if and only if virtual costs are ordered like standard marginal conservation costs θ^{os} (i.e., $V_A(A_{LA}^{SB}) < V_A(A_{LP}^{SB}) < V_A(A_{HA}^{SB}) < V_A(A_{HP}^{SB})$). Since we do not consider classical assumptions such as constant cost increases between types and consider that the two conservation cost parameters are independently distributed, it is worth investigating conditions for the emergence of a separating equilibrium under asymmetric information. We derive the following lemma:

Lemma 4:

Under the assumption $2\Delta\theta^s < \Delta\theta^o$:

- *We have $V_A(A_{LP}^{SB}) < V_A(A_{HA}^{SB})$ and $A_{HA}^{SB} < A_{LP}^{SB}$ if:*

$$\frac{\mu}{1 - \mu} < 1 + \frac{\nu}{(1 - \nu)\mu} \quad (\text{II.16})$$

This inequality is a sufficient condition to ensure the existence of a separating equilibrium between LP and HA-types under asymmetric information.

- We have $V_A(A_{HA}^{SB}) < V_A(A_{HP}^{SB})$ and $A_{HP}^{SB} < A_{HA}^{SB}$ only if:

$$\frac{\nu}{(1-\nu)\mu} < 1 + \frac{\nu + (1-\nu)\mu}{(1-\nu)(1-\mu)} \quad (\text{II.17})$$

*This inequality is a necessary condition to ensure the existence of a separating equilibrium between HA and HP types under asymmetric information.*¹⁵

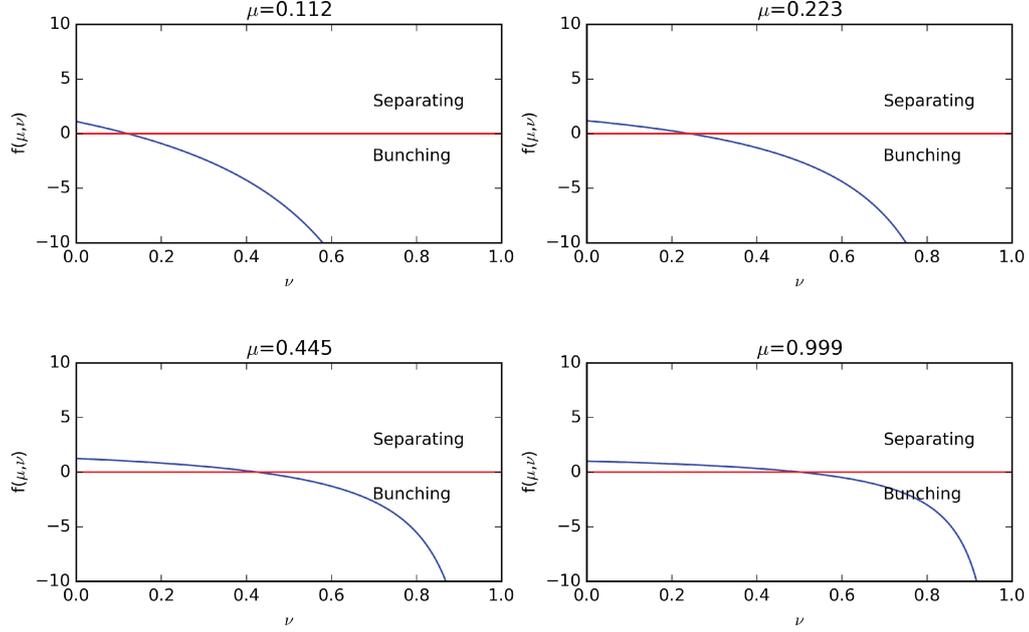
Proof: see Appendix 8.3

It can be analytically showed that a bunching equilibrium between HA- and HP-type agents (i.e., the impossibility to define differentiated contracts) is less likely to emerge when the species absence probability $\mu > 0.5$ and that an increase in μ favours the emergence of a separating equilibrium. Besides, an increase in the proportion of low opportunity cost agents ν favours the emergence of a bunching equilibrium between HA- and HP-type agents (see Figure II.1).

On the contrary, a bunching equilibrium between LP- and HA-type agents cannot emerge when $\mu < 0.5$. An increase in μ favours the emergence of such a bunching equilibrium and an increase in ν favours screening. Moreover, the inequality $\mu > \nu$ always holds when inequality (II.16) is violated.

The performance of comparative statics shows us that inequalities (II.16) and (II.17) can be verified under a set of limited conditions: for high values of μ , $\mu \gg \nu$ leads to violation of inequality (II.16) (see Figure II.2). Thus, if the program targets a very rare species, LP- and HA-type agents are likely to be offered the same contract. On the contrary, $\nu \gg \mu$ leads to the violation of inequality (II.17) especially for low values of μ (see Figure II.1).

Figure II.1: Influence of the species' absence probability μ and the proportion of low opportunity cost agents ν in the landscape on the emergence of a screening equilibria between HA- and HP-type agents



Notes: Here, $f(\mu, \nu) = 1 + \frac{\nu + (1-\nu)\mu}{(1-\nu)(1-\mu)} - \frac{\nu}{(1-\nu)\mu}$ (in blue). A bunching equilibrium occurs between HA- and HP-type agents if $f(\mu, \nu) \leq 0$ and a separating equilibrium occurs between HA- and HP-type agents only if $f(\mu, \nu) \geq 0$. The asymptote $y = 0$ (in red) allows to sort bunching and separating equilibria.

We derive the following proposition:

Proposition 3:

When considering non-constant cost increases between types and independently distributed cost parameters under an AS model, the existence of a fully separating equilibrium is verified under a set of limited conditions on species absence probability and the proportion of low opportunity cost agents in the landscape:

- *When the conservation of a rare species takes place in a weakly production-oriented landscape, it is optimal to offer LP- and HA-type agents the same conservation contract and second best allocations correspond to a mixed solution:*

$$A_{LA}^{SB} > A_{LP}^{SB} = A_{HA}^{SB} > A_{HP}^{SB} \quad (\text{II.18})$$

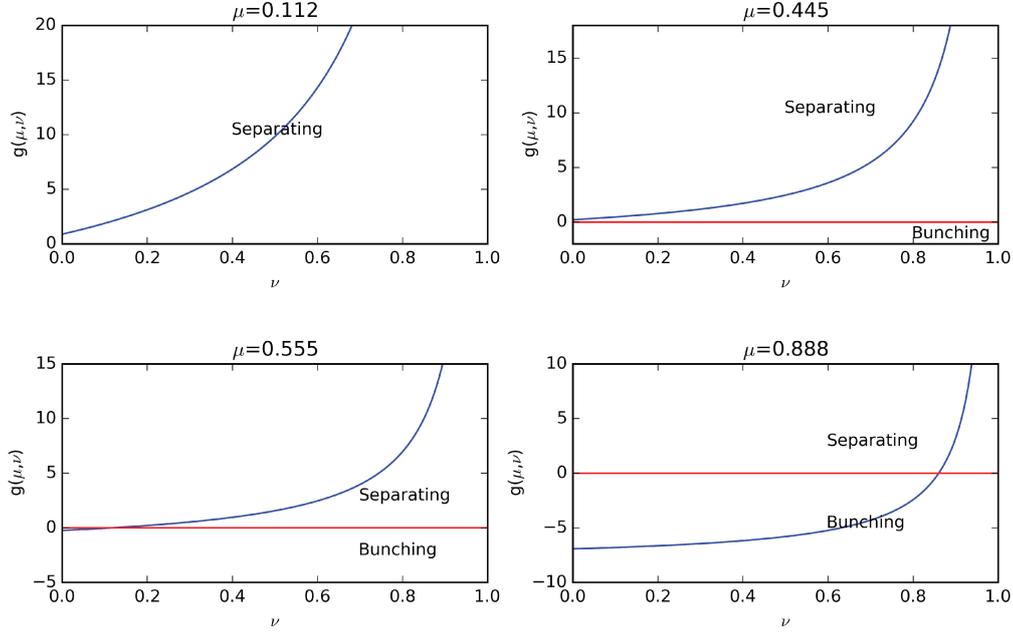
- *When the conservation of a common target species takes place in a highly production-oriented landscape, HA- and HP-type agents are likely to be offered the same conservation contract. In this case, second best allocations correspond to a mixed solution:*

$$A_{LA}^{SB} > A_{LP}^{SB} > A_{HA}^{SB} = A_{HP}^{SB} \quad (\text{II.19})$$

If $\mu \gg \nu$, the considered landscape is poorly suitable for the target species and the effective presence of the target species should be very rare in the landscape. For the principal, opportunities to contract on the effective protection of species individuals are very poor. A high level of conservation cost for an agent has greater chances to be linked to a high level of opportunity cost rather than to the presence of species individuals on his property. In this case, the information rent linked to the potential imitation of the *LP* type is too costly, the area to be conserved by the latter at the second best is distorted downward and it is optimal to offer *LP*- and *HA*-type agents the same conservation contract: $A_{LP}^{SB} = A_{HA}^{SB}$.

By contrast, if $\nu \gg \mu$, the studied region is rather suitable for the target species and a higher level of conservation cost has greater chances to be linked to the effective presence of the species rather than to a high level of opportunity cost. Information costs linked to the potential imitation of *HA*- and *HP*-type agents are high. The area to be conserved by *HA*-type agents is distorted downward and *HA*- and *HP*-type agents are offered the same conservation contract: $A_{HA}^{SB} = A_{HP}^{SB}$.

Figure II.2: Influence of the species' absence probability μ and the proportion of low opportunity cost agents ν in the landscape on the emergence of a screening equilibria between LP- and HA-type agents



Notes: Here, $g(\mu, \nu) = 1 + \frac{\nu}{(1-\nu)\mu} - \frac{\mu}{1-\mu}$ (in blue). A bunching equilibrium occurs between HA- and LP-type agents only if $g(\mu, \nu) \leq 0$ and a separating equilibrium occurs between HA- and LP-type agents if $g(\mu, \nu) \geq 0$. The asymptote $y = 0$ (in red) allows to sort bunching and separating equilibria.

4.2 The common-value model

Program of the conservation planner

Similarly to the AS model, under a CV model and asymmetric information the principal aims at maximizing the difference between expected conservation benefits and expected transfers $E(W)$:

$$\begin{aligned}
 E(W) = & \mu\nu[V(A_{LA}, \theta_A^s) - t_{LA}] + (1 - \mu)\nu[V(A_{LP}, \theta_P^s) - t_{LP}] \\
 & + \mu(1 - \nu)[V(A_{HA}, \theta_A^s) - t_{HA}] + (1 - \mu)(1 - \nu)[V(A_{HP}, \theta_P^s) - t_{HP}] \quad (\text{II.20})
 \end{aligned}$$

The definition of conservation contracts $(A(\theta^{os}), t(\theta^{os}))$ is subject to the agent's individual rationality constraint (see equation (2)) as well as to incentive constraints previously defined (see section 4.1). Again, in this case only R_{LA} , R_{LP} and R_{HA} incen-

tive constraints are binding (see equation (II.13a) to (II.13c)) and only the individual rationality constraint of the HP-type agent is binding (i.e., $R_{HP} = 0$). Thus, under a CV model and asymmetric information, the principal has to solve the following program:

$$\begin{aligned}
\max_{(A(\theta^{os}), R(\theta^{os}))} \quad & EW_{(A(\theta^{os}), R(\theta^{os}))} = \mu\nu[V(A_{LA}, \theta_A^s) - A_{LA} \times (\theta_L^o + \theta_A^s)] \\
& + (1 - \mu)\nu[V(A_{LP}, \theta_P^s) - A_{LP} \times (\theta_L^o + \theta_P^s)] \\
& + \mu(1 - \nu)[V(A_{HA}, \theta_A^s) - A_{HA} \times (\theta_H^o + \theta_A^s)] \\
& + (1 - \mu)(1 - \nu)[V(A_{HP}, \theta_P^s) - A_{HP}(\theta_H^o + \theta_P^s)] \\
& - [\mu\nu \times R_{LA} + (1 - \mu)\nu \times R_{LP} + \mu(1 - \nu) \times R_{HA} + (1 - \mu)(1 - \nu) \times R_{HP}]
\end{aligned} \tag{II.21}$$

We define first order conditions, displayed in equations (II.22a) to (II.22d) hereinbelow.

$$V_A(A_{LA}^{SB}, \theta_A^s) = \theta_L^o + \theta_A^s \tag{II.22a}$$

$$V_A(A_{LP}^{SB}, \theta_P^s) = \theta_L^o + \theta_P^s + \frac{\mu \times \Delta\theta^s}{1 - \mu} \tag{II.22b}$$

$$V_A(A_{HA}^{SB}, \theta_A^s) = \theta_H^o + \theta_A^s + \frac{\nu \times (\Delta\theta^o - \Delta\theta^s)}{(1 - \nu)\mu} \tag{II.22c}$$

$$V_A(A_{HP}^{SB}, \theta_P^s) = \theta_H^o + \theta_P^s + \frac{(\nu + \mu(1 - \nu))\Delta\theta^s}{(1 - \mu)(1 - \nu)} \tag{II.22d}$$

Similarly to the AS model, under a CV model second best conservation contracts $(A^{SB}(\theta^{os}), R(\theta^{os}))$ are defined so that, principal's marginal utility derived from a conserved hectare (i.e., $V_A(A^{SB}(\theta^{os}), \theta^{os})$) equals virtual marginal conservation cost.

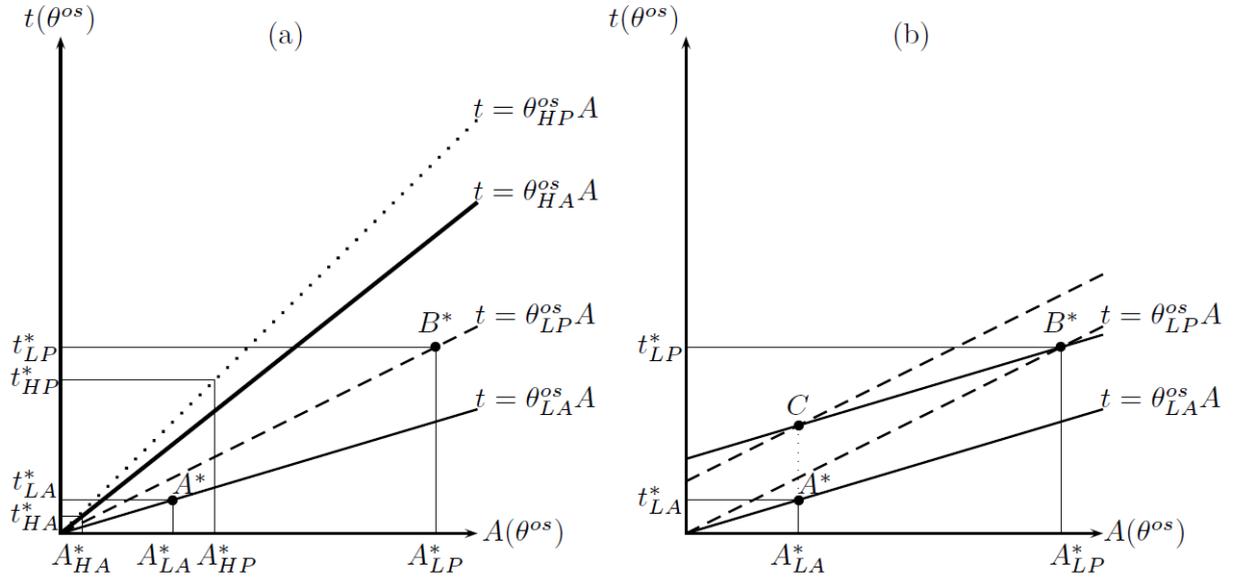
Conditions for the emergence of separating equilibria

In order to be implementable, second-best contracts must satisfy a monotonicity constraint (see e.g., Laffont and Martimort, 2002). Thus, areas to be conserved by each type of agent at the second-best have to increase with cost-efficiency (i.e., decrease with the type index θ^{os}) and we must have:

$$A_{LA}^{SB} \geq A_{LP}^{SB} \geq A_{HA}^{SB} \geq A_{HP}^{SB} \tag{II.23}$$

However, the definition of first-best contracts under the CV model conflicts with this monotonicity constraint. Indeed, as displayed in equation (II.10) efficiency requires the principal to define larger allocations for LP- and HP-type agents compared to allocations defined for LA- and HA-type agents. First-best contracts are therefore not implementable under asymmetric information as explained by Figure II.3. When specifically considering the non-implementability of first-best contracts intended for LP-type agents (i.e., contract B^*) and LA-type agents (i.e., contract A^*) in Figure II.3(b), we can see that the contract B^* is strictly preferred to the contract A^* by the LA-type agent. Contract C generating the same level of utility for the LA-agent and leading the latter to conserve the same area as under first-best contracts can be defined. The LA-type agent is indifferent between these two contracts A^* and C . However, the C contract is not incentive compatible since it would be strictly preferred to the contract B^* by the LP-type agent.

Figure II.3: Non-implementability of first-best contracts under the CV model



Notes: (a) Scheduling of conservation contracts $(A(\theta^{os}), t(\theta^{os}))$ under the CV model. (b) Focus on the non-implementability of first best contracts A^*_{LA} and A^*_{LP} .

Moreover, confronting equations (II.10) and (II.23) shows that first-best and second-best allocations are required to move in opposite directions according to the type in-

dex θ^{os} , which shows that our model illustrates the phenomenon of *non-responsiveness* (Guesnerie and Laffont, 1984).

We investigate the potentialities for the existence of a separating equilibrium under asymmetric information for the CV model. Performing calculations without defining any functional form for the principal's utility function does not allow to conclude. We thus investigate further the potentialities for the existence of a separating equilibrium under asymmetric information by using a Cobb-Douglas function: $V(A(\theta^{os}), \theta^s) = \alpha \times A^\beta \times (\theta^s)^\gamma$, with $\alpha \geq 1$, $0 < \beta < 1$, $\gamma > 0$. We assume that this functional form satisfies the main model assumptions (i.e. $V_A > 0$, $V_{AA} < 0$ and $V_{A\theta^s} > 1$). This function combines a classical ecological benefit function (i.e., $h(A) = \alpha \times A^\beta$) - giving the expected lifetime of a species' population in a patch of size A , as in Drechsler and Wätzold (2001).¹⁶ - with a second term $(\theta^s)^\gamma$. This $(\theta^s)^\gamma$ term renders the twofold conservation objective and the interdependence between habitat suitability provision and species-specific protection.

The value of the γ parameter conveys the additionality of joint provision of effective species-specific protection and habitat suitability services when compared to the provision of the habitat suitability service alone. Appendix 8.5 provides a possible definition of γ values in a context of species-specific conservation contracts across the whole species' range.

Calculations using this functional form allow to derive two kinds of results. First, we show the possibility for the emergence of separating equilibria under asymmetric information for the CV model. We show that the emergence of separating equilibria between *LA*- and *LP*- as well as *LA*- and *HP*-type agents is always possible (see Appendix 8.4). We also show the possibility for the emergence of separating equilibria between *LP*- and *HA*- as well as between *HA*- and *HP*-type agents (see Appendix 8.4). Second, we define conditions for the emergence of such equilibria and derive the following proposition.

¹⁶The ecological benefit function $h(A) = \alpha \times A^\beta$ can be interpreted in two different ways according to Drechsler and Wätzold (2001): 1) it can give the number of species found in a habitat patch of size A , but also 2) it can give the expected lifetime of a species' population in a patch of size A . In the former case, the value of the β parameter depends on the species and on the type and structure of the habitat; in the latter case, the value of the β parameter is inversely related to the temporal variation in the population growth depending itself on the species and habitat quality (Wätzold and Drechsler, 2005).

Proposition 4:

When considering adjacent types of agents in a pairwise manner, individually differentiated contracts can be defined for each of the two types of agent when the bonus provided in terms of species-specific protection by the less cost-efficient type is outweighed by the additional virtual costs related to this type. Thus, separating equilibria emerge if and only if the following conditions are satisfied:

$$- A_{LA}^{SB} > A_{LP}^{SB} \text{ iff } \left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma < \frac{(1-\mu)(\theta_L^o + \theta_P^s) + \mu\Delta\theta^s}{(1-\mu)(\theta_L^o + \theta_A^s)} \quad (22a)$$

$$- A_{LP}^{SB} > A_{HA}^{SB} \text{ iff } \left(\frac{\theta_A^s}{\theta_P^s}\right)^\gamma < \frac{(\theta_H^o + \theta_A^s)(1-\nu)\mu + (\Delta\theta^o - \Delta\theta^s)\nu}{(1-\nu)\mu} \times \frac{(1-\mu)}{(1-\mu)(\theta_L^o + \theta_P^s) + \mu\Delta\theta^s} \quad (22b)$$

$$- A_{HA}^{SB} > A_{HP}^{SB} \text{ iff} \\ \left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma < \frac{(1-\mu)(1-\nu)(\theta_H^o + \theta_P^s) + (\nu + (1-\nu)\mu)\Delta\theta^s}{(1-\mu)(1-\nu)} \times \frac{(1-\nu)\mu}{(\theta_H^o + \theta_A^s)(1-\nu)\mu + (\Delta\theta^o - \Delta\theta^s)\nu} \quad (22c)$$

$$- A_{LA}^{SB} > A_{HP}^{SB} \text{ iff } \left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma < \frac{(1-\mu)(1-\nu)(\theta_H^o + \theta_P^s) + (\nu + (1-\nu)\mu)\Delta\theta^s}{(1-\mu)(1-\nu)(\theta_L^o + \theta_A^s)} \quad (22d)$$

Proof: see Appendix 8.4

The $\left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma$ term in the left-hand side of equations (22a), (22c) and (22d) corresponds to the ratio between the principal's marginal utility of one hectare conserved by an agent of type θ_{LP}^{os} (θ_{HP}^{os} for equations (22c) and (22d)) and the marginal utility of one hectare conserved by an agent of type θ_{LA}^{os} (θ_{HA}^{os} and θ_{LA}^{os} for equations (22c) and (22d) respectively). This ratio thus allows to quantify the bonus provided by the conservation of one hectare by θ_{LP}^{os} - and θ_{HP}^{os} -type agents compared to a θ_{LA}^{os} and θ_{HA}^{os} -type agents in terms of conservation benefits. Differentiated contracts can be defined between both types of agent (i.e., LP/LA , HP/HA and HP/LA for equations (22a), (22c) and (22d) respectively) when this bonus is outweighed by the ratio of virtual costs. It can be analytically showed that $\left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma$ increases with γ , whereas $\left(\frac{\theta_A^s}{\theta_P^s}\right)^\gamma$ decreases with γ . Thus, when the level of additionality of joint provision of effective species-specific protection and habitat suitability services increases, the bonus provided by the conservation of one hectare by LP - and HP -type agents compared to LA - and HA -type agents increases. This favours the emergence of LP/HA separating equilibria as well as the emergence of LA/LP , HA/HP and LA/HP bunching equilibria.

We differentiate the right-hand side (RHS) of equations (22a) to (22d) according to the several model parameters (i.e., μ , ν , $\Delta\theta^o$ and $\Delta\theta^o - \Delta\theta^s$) and perform comparative statics to determine the impact of an increase in these parameters on the type of emerging equilibria (i.e., bunching or screening) between adjacent types of agents. Results are summarized and displayed in Table II.2.

We cannot determine the impact of an increase in $\Delta\theta^s$ on the type of emerging equilibria between adjacent types of agents. Indeed an increase in $\Delta\theta^s$ has an ambiguous influence on the four studied equilibria since both right and left hand sides of equations (22a), (22c) and (22d) are increasing according to $\Delta\theta^s$, and left and right hand sides of equation (22b) are decreasing according to $\Delta\theta^s$.

Thus, according to analytical results at least five different kinds of “global” equilibria can emerge (i.e., menus of contracts): the individual violation of inequalities (22a) to (22d) leads to the emergence of mixed equilibria, the emergence of fully separating and full bunching equilibria has to be considered as well as eventual combinations of mixed equilibria.

5 Numerical simulations at the landscape scale

Now we aim at comparing the performance of AS and CV models in terms of differentiation of contracts and cost-effectiveness when implemented at the landscape scale. To do this, we simulate the implementation of conservation contracts derived from both types of model in two hypothetical landscapes built based on two case studies (see Section 5.2).

5.1 Landscape structure

We consider the implementation of a conservation program in a hypothetical landscape corresponding to an average size French forest massif (approx. 4,950 ha) divided into 2,750 individual cells of 1.8 ha.¹⁷ We assume that each individual cell corresponds to

¹⁷According to the results of the 2012 Agreste survey on the structure of the French private forest, the average size of a continuous forest parcel belonging to a unique private owner in France equals 1.8 ha. Thus, we consider that such a forest parcel corresponds to an individual forest management decision unit.

Table II.2: Influence of parameters on the type of emerging equilibria

Parameter	Impact	Type of equilibrium (separating vs. bunching)
μ	$\mu \nearrow$: RHS eq. (22a) \nearrow	<i>LA/LP</i> separating
	$\mu \nearrow$: RHS eq. (22b) \searrow	<i>LP/HA</i> bunching
	$\mu \nearrow$: RHS eq. (22c) \nearrow	<i>HA/HP</i> separating
	$\mu \nearrow$: RHS eq. (22d) \nearrow	<i>LA/HP</i> separating
ν	$\nu \nearrow$: RHS eq. (22a) \rightarrow	no impact on <i>LA/LP</i>
	$\nu \nearrow$: RHS eq. (22b) \nearrow	<i>LP/HA</i> separating
	$\nu \nearrow$: RHS eq. (22c) ?	ambiguous impact on <i>HA/HP</i>
	$\nu \nearrow$: RHS eq. (22d) \nearrow	<i>LA/HP</i> separating
$\Delta\theta^o$	$\Delta\theta^o \nearrow$: RHS eq. (22a) \rightarrow	no impact on <i>LA/LP</i> eq.
	$\Delta\theta^o \nearrow$: RHS eq. (22b) \nearrow	<i>LP/HA</i> separating eq.
	$\Delta\theta^o \nearrow$: RHS eq. (22c) ?	ambiguous impact on <i>HA/HP</i> eq.
	$\Delta\theta^o \nearrow$: RHS eq.(22d) \nearrow	<i>LA/HP</i> separating eq.
$\Delta\theta^o - \Delta\theta^s$	$\Delta\theta^o - \Delta\theta^s \nearrow$: RHS eq. (22a) \rightarrow	no impact on <i>LA/LP</i> eq.
	$\Delta\theta^o - \Delta\theta^s \nearrow$: RHS eq. (22b) \nearrow	<i>LP/HA</i> separating eq.
	$\Delta\theta^o - \Delta\theta^s \nearrow$: RHS eq. (22c) \searrow	<i>HA/HP</i> bunching eq.
	$\Delta\theta^o - \Delta\theta^s \nearrow$: RHS eq. (22d) \rightarrow	no impact on <i>LA/HP</i> eq.

Notes: For the record, μ corresponds to species absence probability ; ν corresponds to the proportion of low opportunity cost agents ; $\Delta\theta^s$ and $\Delta\theta^o$ correspond to the difference between high and low levels of protection and opportunity cost respectively.

an individual property and assume a single individual owner for each cell.

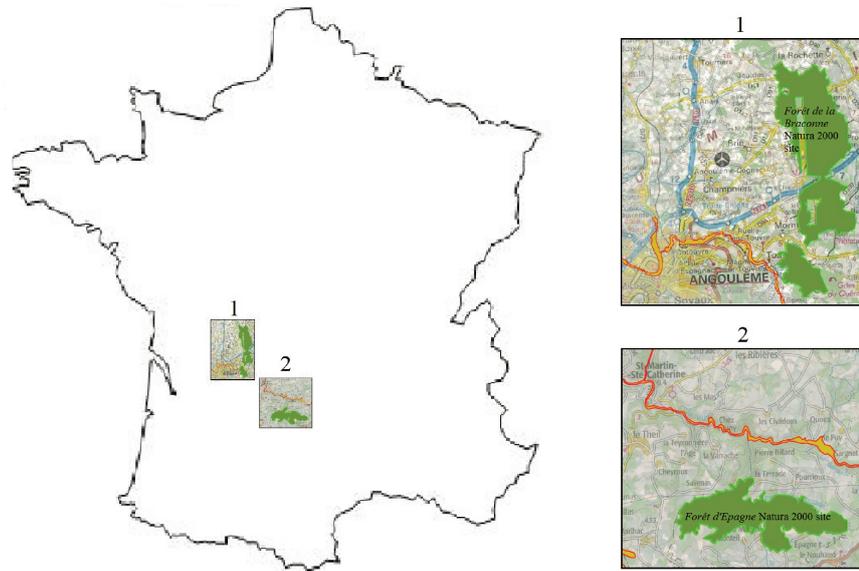
Conservation costs and protection costs (i.e., species' presence) are randomly distributed across the landscape with no spatial correlation. Assuming spatially uncorrelated conservation costs in the simulation is not problematic since we consider a static model and do not consider any spatially interdependent conservation benefits.

5.2 Case studies and definition of values of simulation parameters

We consider a conservation program, which targets a herbaceous vegetal species. This program requires forest owners to 1) set-up uneven-aged stands in their property in order to provide suitable habitat¹⁸ and 2) implement species-specific protection measures. In

¹⁸As mentioned in the management plan of the Natura 2000 site *La Forêt d'Epagne*, managing a forest stand in an irregular manner (i.e., uneven-aged stands) allows to preserve biodiversity and enhances

Figure II.4: Geographical locations of case studies



(1) *Forêt de la Braconne*, (2) *Forêt d'Espagne*

absence of the species in the property, protection measures only consist in conducting regular monitoring to detect the arrival of individuals. Protection in presence of the species implies both monitoring and active protection of species individuals which are present in the property. Following Mewes et al. (2015), we consider opportunity costs of conservation (i.e., θ^o) as the sum of a) direct costs due to the implementation of silvicultural operations when shifting into uneven-aged forest stands, b) income losses in terms of timber production and c) additional administrative costs (i.e., opportunity cost of time).

As previously mentioned, we define landscape characteristics and simulation parameter values based on two case studies: *La Forêt d'Espagne* forest massif located in the Limousin region and *La Forêt de la Braconne* forest massif located in the Poitou-Charentes region (see Figure II.4). Both forest massifs belong to a Natura 2000 site and have been chosen because they are located in regions differing in terms of local timber production contexts.

Estimates for the cost of silvicultural operations when shifting to uneven-aged stands are provided by Natura 2000 site-specific management plans for both case studies. The carrying capacity.

average cost of shifting into uneven-aged stands for the *Forêt d'Epagne* case study equals €2,100/ha for five years, that is €420/ha/year. The cost of this shift equals €577/ha/year for the *Forêt de la Braconne* case study.¹⁹

Cost estimates provided by Natura 2000 management plans do not consider opportunity costs in terms of losses from timber production. Based on regional timber productivity data (IGN 2013 inventory) and timber prices (standing timber prices index, *Forêt Privée Française*) we calculate income losses while considering that conservation contracts' implementation triggers a loss of 10% in terms of timber productivity, that is €67/ha/year and €64/ha/year for the *Forêt d'Epagne* and *Forêt de la Braconne* case studies respectively. The proportion of agents showing a high level of opportunity cost ($1 - \nu$) is approximated by the proportion of the regional forest area based on plantations (i.e., 25% in Limousin, 9% in Poitou-Charentes, IGN 2013 inventory).

Additional administrative costs incurred by forest owners are defined based on estimates found in the *Forêt de la Braconne* management plan. It is assumed that participants to the conservation program have to attend half-day public meetings twice a year and spend a whole day's work (8 hours) on administrative work related to the conservation contract's implementation. We assume that average hourly opportunity cost of time spent on administrative work at least equals the minimum French hourly wage (i.e., €9.61/hour). Then, opportunity cost of time equals €154/year, that is €85/ha/year when averaged over 1.8 ha.

Finally, species-specific protection costs (i.e., θ^s) are defined based on cost estimates provided in the *Forêt de la Braconne* management plan. Monitoring cost is approximated by the cost of an ecological diagnosis, being equal to €20/ha/year. Active protection of species individuals is approximated by the protection of plantation seedlings, whose cost equals €400/ha/year. Table II.3 sums-up the various values defined for the conservation cost parameters.

¹⁹This difference in terms of cost of silvicultural operations can be explained by the fact that the *Forêt de la Braconne* Natura 2000 site contains much more private forest properties and valuable forests (e.g., even-aged oak and conifer plantations) than the *Forêt d'Epagne*, in which forest is managed mainly as a coppice.

Table II.3: Values of conservation cost parameters (in €/ha/yr)

<i>Forêt d'Epagne</i>		<i>Forêt de la Braconne</i>	
Protection cost θ^s	Opportunity cost θ^o	Protection cost θ^s	Opportunity cost θ^o
$\theta_A^s = 20$	$\theta_L^o = 85$	$\theta_A^s = 20$	$\theta_L^o = 85$
$\theta_P^s = 420$	$\theta_H^o = 572$	$\theta_P^s = 420$	$\theta_H^o = 726$

5.3 Conservation scenarios

Parameters values considered in the simulations are presented in Table II.4. A unique combination of parameter values allows to define a conservation scenario. For each of the two case studies, we define 756 scenarios: 108 scenarios are defined for the implementation of contracts derived from an AS model; 648 scenarios are defined for the implementation of contracts derived from a CV model.

Table II.4: Simulation scenarios

Parameter	Description	Considered range
γ	Additionality of species protection	{0.5; 0.7; 0.9; 1.5; 2; 3}
B	Conservation budget in K€	{50; 100; 150}
μ	Species' absence probability	{0.9; 0.99; 0.999}
α	Multiplication factor (principal's utility function)	{1; 2; 3}
β	Population lifetime-Area parameter	{0.1; 0.3; 0.5; 0.7}

We consider 300 iterations (i.e., different landscapes) for each conservation scenario, which allows us to account for the randomness of the hypothetical landscape. Concave and convex principal's utility functions according to θ^s are considered (i.e., low/high additionality of species-specific protection). We also study the influence of budget size (B) and of species rarity (μ) on conservation programs' cost-effectiveness. We analytically showed that the α and β parameters do not have any influence on the nature of the emerging equilibrium (i.e., separating or bunching; see section 4). However allocation size is highly sensitive to the value of these parameters. Therefore, we also investigate the influence of these parameters on conservation programs' cost-effectiveness.

5.4 Conservation outcome

For both types of model and for each scenario we measure the performance of a conservation program according to its i) total cost, ii) total information cost, iii) total area conserved, iv) total area conserved with presence of species individuals. We use these

four indicators to calculate the cost-effectiveness of contracts derived from both types of model.

5.5 Results

Differentiation of conservation contracts

Simulation results show that basing the definition of conservation contracts on an AS model results in the invariable emergence of the following mixed equilibrium: $A_{LA}^{SB} > A_{LP}^{SB} = A_{HA}^{SB} > A_{HP}^{SB}$. In this case, agents providing effective species protection at low opportunity cost (i.e., *LP*-type agents) and agents providing no effective species protection but habitat suitability at high opportunity cost (i.e., *HA*-type agents) are offered the same contract. Thus it is impossible to target agents providing effective protection services at the lowest cost. Conservation contracts are “prioritized” towards the most cost-efficient agent-type (i.e., *LA*-type agents), which allows to reduce information costs (see next section 5.5).

Results of numerical simulations confirm and extend analytical results by showing the possibility to define differentiated contracts under a CV model. We show that an increase in γ leads to the successive emergence of at most 5 different kinds of equilibria according to a stable pattern,²⁰ as summarized in Table II.5.

Thus, defining conservation contracts based on a CV model when species-specific protection is poorly additional (i.e., low γ values) leads to the emergence of an equilibrium of type (Eq1). *LP*- and *HA*-type agents are bunched and offered the same contract. Here, CV contracts show similar screening performances to the one obtained with an AS model for both case studies. In that case, basing conservation contracts definition on a CV model does not bring any advantage in terms of contract differentiation.

When higher γ values are considered *HA*- and *LP*-type agents are no longer bunched. It can be observed that basing conservation contracts’ definition on a CV model is particularly interesting in the case of intermediate additionality of species-specific protection. A fully separating equilibrium (Eq2) emerges from $\gamma = 0.83$ and from $\gamma = 1.27$ for the *Forêt de la Braconne* and *Forêt d’Epagne* case studies respectively (with $\mu = 0.99$,

²⁰As revealed through additional numerical comparative statics investigating the impact of individual and joint variations in the value of γ , $\Delta\theta^s$ and $\Delta\theta^o$ parameters on the emergence of the different kinds of equilibrium. However, we find that when considering parameter values defined based on the two case studies the equilibrium of type (Eq3) never emerges.

$\beta = 0.1$ and $\alpha = 1$). In this case, individually differentiated contracts can be defined for each type of agent. An equilibrium of type (Eq4) emerges from $\gamma = 1.63$ and from $\gamma = 1.72$ for the *Forêt de la Braconne* and *Forêt d'Epagne* case studies respectively (with $\mu = 0.99$, $\beta = 0.1$ and $\alpha = 1$). It can be noticed in Figure II.5 that the interval of γ values leading to the emergence of such “virtuous” equilibria is wider for the *Forêt de la Braconne* compared to the *Forêt d'Epagne*.

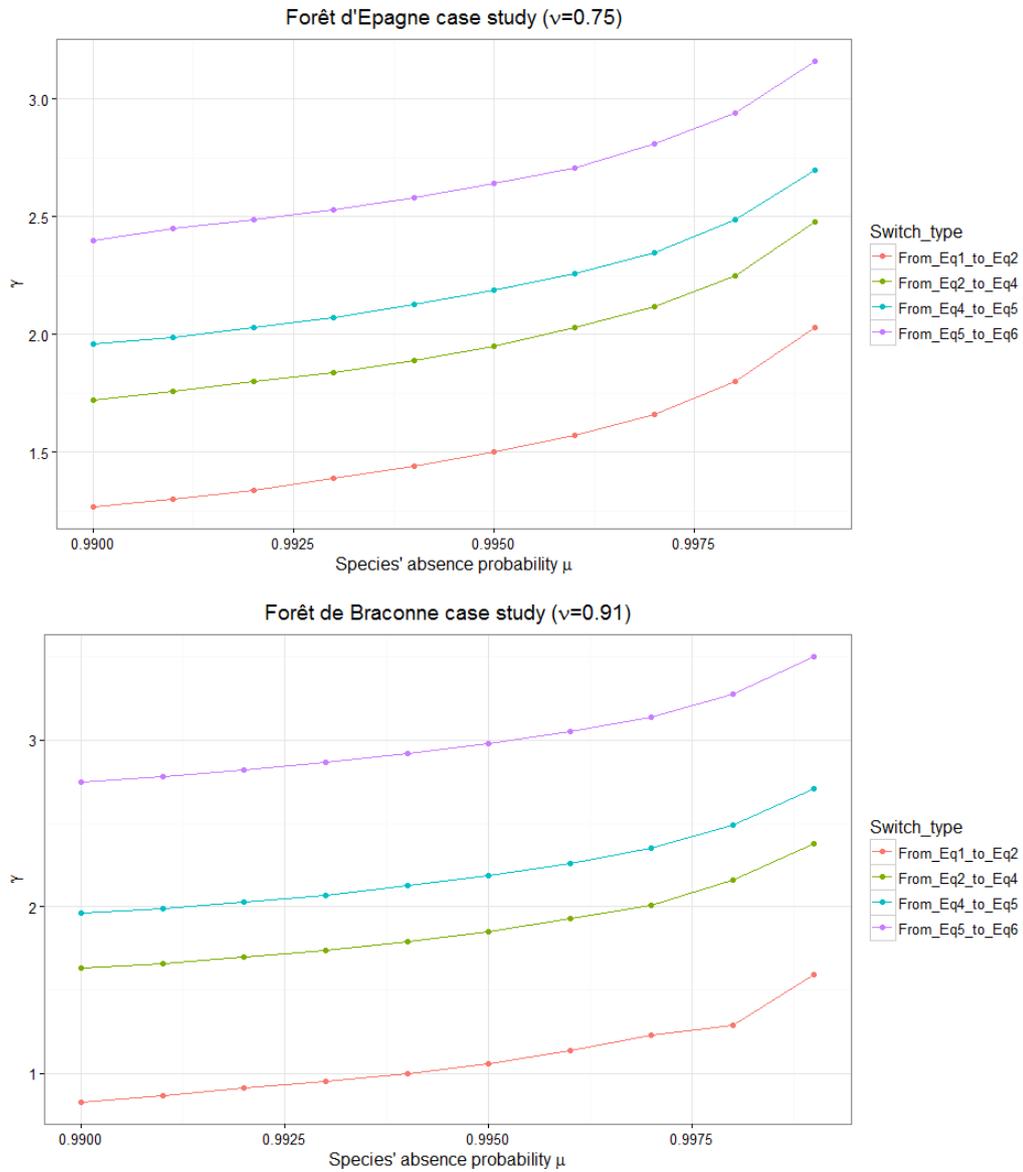
Table II.5: Impact of an increase in γ values on the type of emerging equilibria

Type of equilibrium	
(Eq1):	$A_{LA}^{SB} > A_{LP}^{SB} = A_{HA}^{SB} > A_{HP}^{SB}$
(Eq2):	$A_{LA}^{SB} > A_{LP}^{SB} > A_{HA}^{SB} > A_{HP}^{SB}$
(Eq3):	$A_{LA}^{SB} = A_{LP}^{SB} > A_{HA}^{SB} > A_{HP}^{SB}$ (low $\Delta\theta_o$)
or (Eq4):	$A_{LA}^{SB} > A_{LP}^{SB} > A_{HA}^{SB} = A_{HP}^{SB}$ (high $\Delta\theta_o$)
(Eq5):	$A_{LA}^{SB} = A_{LP}^{SB} > A_{HA}^{SB} = A_{HP}^{SB}$
(Eq6):	$A_{LA}^{SB} = A_{LP}^{SB} = A_{HA}^{SB} = A_{HP}^{SB}$

Basing conservation contracts' definition on a CV model is no longer interesting when species-specific protection is highly additional (i.e., high γ values). For both case studies, agents providing effective species protection (i.e., *LP*- and *HP*-type agents) cannot be distinguished from others from $\gamma = 1.96$ (with $\mu = 0.99$, $\beta = 0.1$ and $\alpha = 1$). The emerging equilibrium is equivalent to screening agents on a opportunity-cost-basis only (i.e., (Eq5)) over a first phase. A further increase in γ leads to a fully bunching equilibrium (i.e., (Eq6)) from $\gamma = 2.75$ and from $\gamma = 2.4$ for the *Forêt de la Braconne* and *Forêt d'Epagne* case studies respectively .

Finally, it can be interesting to observe the influence of species' rarity on the performance of CV contracts in terms of contract differentiation. As species' rarity (i.e., μ) increases, the emergence of the different kinds of equilibria is shifted towards higher γ values (see Figure II.5). As mentioned in the analytical part of the chapter, the value of α and β parameters has no influence on the nature of the emerging equilibrium, therefore results displayed in Figure II.5 hold for all considered α and β values.

Figure II.5: γ threshold values for equilibrium switches

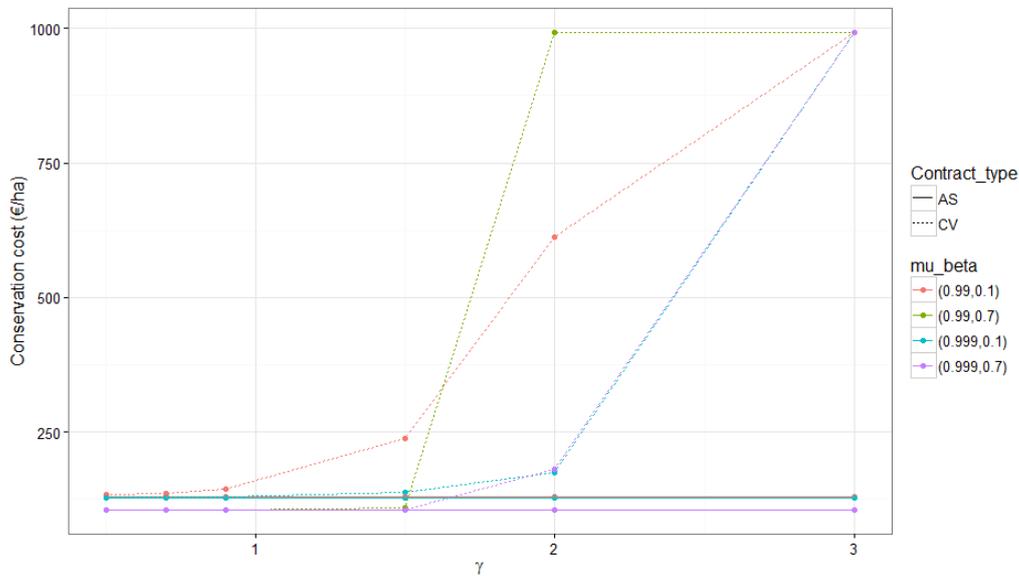


Cost-effectiveness of conservation contracts

Figures II.6 and II.7 display the relative cost-effectiveness performance of AS and CV contracts regarding conservation cost and information cost respectively. They show that the definition of conservation contracts based on an AS model allows to minimize the cost of a conserved hectare (Fig. II.6) as well as the share of conservation cost due to information cost (Fig. II.7) when compared with a CV model.

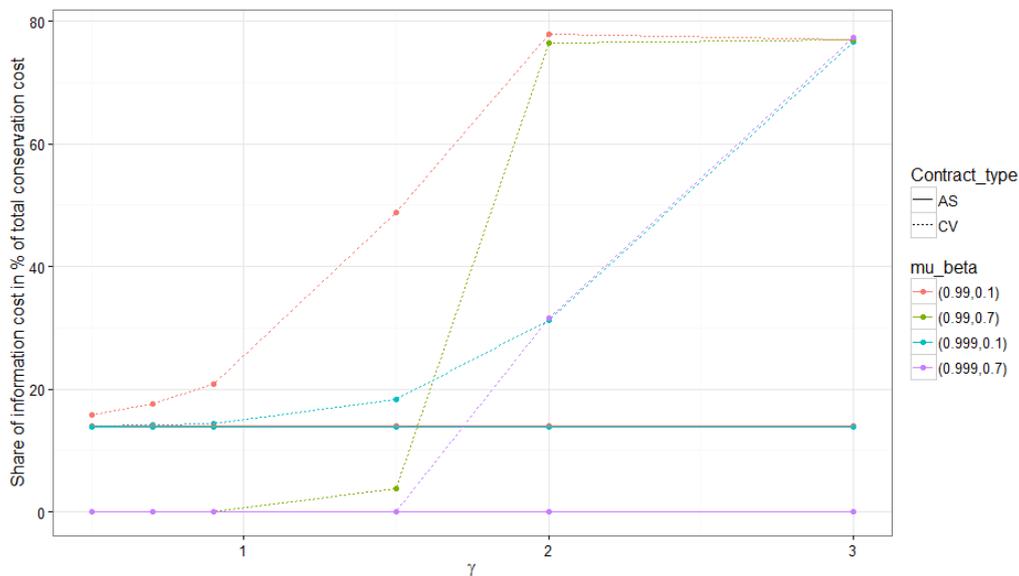
When species-specific protection is poorly additional (i.e., low γ values) CV contracts

Figure II.6: Relative cost-effectiveness performance of CV and AS contracts - Conservation cost



Notes: For readability reasons results are displayed for combinations of two values of μ and β parameters ($\beta \in \{0.1; 0.7\}$ and $\mu \in \{0.99; 0.999\}$). This figure displays results for the *Epagne* case study; same trends are observed for the *Braconne* case study.

Figure II.7: Relative cost-effectiveness performance of CV and AS contracts - Level of information cost



Notes: For readability reasons results are displayed for combinations of two values of μ and β parameters ($\beta \in \{0.1; 0.7\}$ and $\mu \in \{0.99; 0.999\}$). This figure displays results for the *Epagne* case study; same trends are observed for the *Braconne* case study.

show similar cost-effectiveness to AS contracts. As shown by Figures II.6 and II.7 respectively, under CV contracts the cost of one conserved hectare is kept low as well as information cost despite an increase in γ (for $\gamma < 1.5$). Given this result and the result regarding the type of equilibrium emerging under a CV model for low additionality of protection, the interest of defining contracts based on a CV model in this context can be questioned. In the case of low additionality of species protection it could be more efficient to screen agents on an opportunity cost basis only.

As for contract differentiation, basing conservation contracts' definition on a CV model is particularly cost-effective in the case of intermediate additionality of species-specific protection. This is particularly true in the case of weakly production-oriented (higher ν values) and low-productivity landscapes (low $\Delta\theta^o$ values). For instance, an increase in γ from $\gamma = 0.9$ to $\gamma = 1.5$ leads to a significant increase of information costs from 21% to 49% of total conservation cost for the *Forêt d'Epagne* case study (with $\mu = 0.99$, $\beta = 0.1$ and $\alpha = 1$). However, the cost of one conserved hectare is held relatively low: conservation cost increases from €144/ha to €237/ha.

When species-specific protection is highly additional (i.e., high γ values), CV contracts show poor cost-effectiveness. Information costs associated to the implementation of CV contracts are very high (see Figure II.7 e.g., close to 80% of total conservation costs) and the cost of one conserved hectare rises sharply with γ for $\gamma \geq 1.5$ (see Figure II.6). This result concurs with results regarding contract differentiation and suggests that, for considered case studies and in a context of high additionality of species-specific protection, cost-effective species-specific conservation cannot be based on incentive conservation contracts. Thus, in a context of high additionality of species-specific protection, basing conservation on another approach such as publicly managed natural protected areas could be more efficient.

Conservation cost and information cost increase with and increase in α . On the contrary, conservation and information cost decrease with an increase in β . These results can be related to the fact that allocations (i.e., area to be conserved) increase with α but decrease with β (see for instance Appendix 8.4).

The level of available conservation budget available has no influence on the relative cost-effectiveness of AS and CV contracts. Indeed, AS contracts require the conservation of very small areas and the budget is never exhausted. On the contrary, CV contracts

lead to budget's exhaustion for intermediate and high level of additionality of species-specific protection (i.e., $\gamma \leq 1.5$).

Finally, it can be interesting to observe the influence of species' rarity on the cost-effectiveness of CV contracts. It can be observed from Figure II.6 and II.7 that both the cost of one conserved hectare as well as the share of total conservation cost due to information costs are kept low for higher γ values when CV contracts are applied to the conservation of a rarer species (i.e., characterized by a higher absence probability μ). Such results highlight the specific interest of CV contracts for the conservation of very rare species.

6 Discussion and Conclusion

In this chapter we investigated the possibility to define differentiated contracts for species-specific conservation when both conservation costs and benefits are unobservable and heterogeneous among landowners. In this context, we analytically and numerically demonstrate the possibility to define differentiated conservation contracts under a *common-value* model (CV). Differentiated contracts can be defined despite a *non-responsiveness* phenomenon known to usually lead to bunching equilibria (Guesnerie and Laffont, 1984). Here we extend the result of Morand and Thomas (2006) who demonstrated the non-systematic emergence of fully bunching equilibria for the definition of contracts for the delegated provision of an indivisible good.

Analytical and simulation results show that second-best contracts defined under a CV model are always strongly or weakly inefficient²¹ with respect to first-best contracts. Indeed, agents providing effective species-specific protection (i.e., *LP*- and *HP*-type agents) are granted areas for conservation, which are lower (strong inefficiency) or equal (weak inefficiency) to areas to be conserved by agents providing no effective species protection (i.e., *LA*- and *HA*-type agents). Therefore, conservation contracts cannot be prioritized towards agents providing effective species protection services.

Conservation contracts defined based on a classic adverse selection model (AS) are always strongly inefficient with respect to first-best contracts defined under a CV model. This especially holds when the conservation of a rare species is at stake (i.e., low μ). The

²¹As defined by Morand and Thomas (2006).

results derived from numerical simulations show that the definition of fully differentiated contracts is not possible under an AS model, while fully differentiated contracts can be defined under a CV model.

We show that the interest of basing the definition of conservation contracts on a CV model strongly depends on the level of additionality of species-specific protection. While it would be very efficient to define species-specific conservation contracts based on a CV model (compared to an AS model) in the case of intermediate additionality of species protection, it would be strongly inefficient to do so in the case of low additionality of species protection. We also show that species-specific conservation should not rely on incentive conservation contracts at all in the case of high additionality of species protection.

Our results are in line with the reality of biodiversity conservation. Indeed, the conservation of very vulnerable and threatened species is not traditionally based on incentive tools and rather relies on a mix of *in situ* and *ex situ* conservation strategies conducted by a public conservation agency. When dealing with biodiversity conservation under climate change, Heller and Zavaleta (2009) underline the fact that protection of historical species in their current distribution would require intense management actions and would “align best with a fixed-reserve approach”. Such a conservation approach would be increasingly costly and challenging over time since species would become more and more threatened by global changes and would likely be entrusted to public conservation agencies.

To sum up, our results point to the significant differences between performances of AS and CV models both in terms cost-effectiveness and contracts differentiation. This shows the need to define species-specific conservation instruments while accounting for conservation objectives as well as for local economic and ecological contexts, and this to ensure cost-effectiveness.

Given this prevailing influence of the γ parameter on the cost-effectiveness of CV contracts, it would have been interesting to estimate its value for the two considered case studies. In this work, available data did not allow us to do so. The application of our model to a real species-specific conservation program in future research would allow to estimate this γ value and formulate more specific policy recommendations. The estimation of the value of the β parameter of the principal's utility function would

require to get precise information about metapopulation dynamics of the target species. In addition to information on species-specific protection cost and the total area conserved in the conservation program, the estimation of the value of the γ parameter would require to get information on the marginal substitution rate between habitat suitability only and effective species-specific protection for the conservation agency.

7 Acknowledgments

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8 Appendix

8.1 First best allocations' definition under the common-value model

Proof - Lemma 1

We follow Laffont and Martimort (2002) and differentiate $W_A(A(\theta^{os}), \theta^o, \theta^s)$ at the optimum $A^*(\theta^{os})$

$$dW_{A^*} = \frac{\partial W_{A^*}}{\partial A^*} dA^* + \frac{\partial W_{A^*}}{\partial \theta^o} d\theta^o + \frac{\partial W_{A^*}}{\partial \theta^s} d\theta^s \quad (A1)$$

$$dW_{A^*} = \frac{\partial W_{A^*}}{\partial V_{A^*}} \left(\frac{\partial V_{A^*}}{\partial A^*} dA^* + \frac{\partial V_{A^*}}{\partial \theta^o} d\theta^o + \frac{\partial V_{A^*}}{\partial \theta^s} d\theta^s \right) \quad (A2)$$

But $W_A(A(\theta^{os}), \theta^o, \theta^s) = V(A(\theta^{os}), \theta^s) - A(\theta^{os}) \times (\theta^o + \theta^s)$ (A3) and $\frac{\partial W_{A^*}}{\partial V_{A^*}} = 1$ (A4)

Thus we can write:

$$dW_{A^*} = \frac{\partial V_{A^*}}{\partial A^*} dA^* + \frac{\partial V_{A^*}}{\partial \theta^o} d\theta^o + \frac{\partial V_{A^*}}{\partial \theta^s} d\theta^s = dV_A(A^*, \theta^s) \quad (A5)$$

Yet, at the optimum we have $V_A(A^*(\theta^{os}), \theta^s) = \theta^o + \theta^s$ and by assumption we have $V_{A\theta^s} > 1$ et $V_{AA} \leq 0$. Thus we can write:

$$V_{AA} dA^* + V_{A\theta^s} d\theta^s + V_{A\theta^o} d\theta^o = d\theta^s + d\theta^o \quad (A6)$$

We assume $V_{A\theta^o} = 0$

Thus we have: $V_{AA} \frac{dA^*}{d\theta^s} + V_{A\theta^s} \frac{d\theta^s}{d\theta^s} = 1 + \frac{d\theta^o}{d\theta^s}$ (A7)

By assumption θ^s and θ^o are independently drawn, thus we have: $\frac{d\theta^o}{d\theta^s} = 0$

Thus: $\frac{dA^*}{d\theta^s} = \frac{1 - V_{A\theta^s}}{V_{AA}} > 0$ (A8)

Proof - Lemma 2

Similarly to the proof of **Lemma 1**, we can write:

$$dW_{A^*} = \frac{\partial V_{A^*}}{\partial A^*} dA^* + \frac{\partial V_{A^*}}{\partial \theta^o} d\theta^o + \frac{\partial V_{A^*}}{\partial \theta^s} d\theta^s = dV_A(A^*(\theta^{os}), \theta^s) \quad (A9)$$

Yet, at the optimum we have $V_A(A^*(\theta^{os}), \theta^s) = \theta^o + \theta^s$ and by assumption we have $V_{A\theta^s} > 1$ et $V_{AA} \leq 0$. Thus we can write:

$$dV_A(A^*(\theta^{os}), \theta^s) = \frac{\partial V_A}{\partial A} dA^* + \frac{\partial V_A}{\partial \theta^s} d\theta^s + \frac{\partial V_A}{\partial \theta^o} d\theta^o \quad (A10)$$

$$\iff V_{AA} dA^* + V_{A\theta^s} d\theta^s + V_{A\theta^o} d\theta^o = 0 \times dA^* + d\theta^s + d\theta^o \quad (A11)$$

We assume $V_{A\theta^o} = 0$

Thus we have: $V_{AA} \frac{dA^*}{d\theta^o} + V_{A\theta^s} \frac{d\theta^o}{d\theta^s} = \frac{d\theta^s}{d\theta^o} + 1$ (A12)

We assume that θ^s and θ^o are independently drawn, thus we have: $\frac{d\theta^o}{d\theta^s} = 0$

Thus: $\frac{dA^*}{d\theta^o} = \frac{1}{V_{AA}} < 0$ (A13)

Proof - Lemma 3

In order to determine the parameter (among θ^s and θ^o), whose effect dominates for the definition of first best allocations we calculate:

$$\frac{dA^*}{d\theta^s} \leq \frac{dA^*}{d\theta^o} \Leftrightarrow \frac{1-V_{A\theta^s}}{V_{AA}} \leq \frac{1}{V_{AA}} \quad (A14)$$

$$\Leftrightarrow \frac{1-V_{A\theta^s}}{V_{AA}} - \frac{1}{V_{AA}} \leq 0 \quad (A15)$$

$$\Leftrightarrow -\frac{V_{A\theta^s}}{V_{AA}} \leq 0 \quad (A16)$$

Moreover, we have: $V_{AA} \leq 0$ and $V_{A\theta^s} > 1$

Therefore we have : $\frac{dA^*}{d\theta^s} - \frac{dA^*}{d\theta^o} = -\frac{V_{A\theta^s}}{V_{AA}} \geq 0$ (A17)

Thus: $\frac{dA^*}{d\theta^s} - \frac{dA^*}{d\theta^o} = -\frac{V_{A\theta^s}}{V_{AA}} \geq 0$ (A18)

8.2 Optimal allocations under asymmetric information

Incentive constraints and information rents

Equation (IC) hereinafter describes the incentive constraint of an agent of type θ^{os} .

Thus, a θ^{os} -type landowner is better off when contracting truthfully if and only if:

$$R(\theta^{os}) = t(\theta^{os}) - A(\theta^{os}) \times \theta^{os} \geq R(\widehat{\theta^{os}}) = t(\widehat{\theta^{os}}) - A(\widehat{\theta^{os}}) \times \theta^{os} \quad (IC)$$

The twelve following incentives constraints are considered in our problem:

$$\begin{array}{ll} R_{LA} \geq R_{LP} + A_{LP} \times \Delta\theta^s & (IC_{LA/LP}) \\ R_{LA} \geq R_{HA} + A_{HA} \times \Delta\theta^o & (IC_{LA/HA}) \\ R_{LA} \geq R_{HP} + A_{HP} \times (\Delta\theta^o + \Delta\theta^s) & (IC_{LA/HP}) \\ R_{LP} \geq R_{LA} - A_{LA} \times \Delta\theta^s & (IC_{LP/LA}) \\ R_{LP} \geq R_{HA} + A_{HA} \times (\Delta\theta^o - \Delta\theta^s) & (IC_{LP/HA}) \\ R_{LP} \geq R_{HP} + A_{HP} \times \Delta\theta^o & (IC_{LP/HP}) \\ R_{HA} \geq R_{LA} - A_{LA} \times \Delta\theta^o & (IC_{HA/LA}) \\ R_{HA} \geq R_{LP} + A_{LP} \times (\Delta\theta^s - \Delta\theta^o) & (IC_{HA/LP}) \\ R_{HA} \geq R_{HP} + A_{HP} \times \Delta\theta^s & (IC_{HA/HP}) \\ R_{HP} \geq R_{LA} - A_{LA} \times (\Delta\theta^o + \Delta\theta^s) & (IC_{HP/LA}) \\ R_{HP} \geq R_{LP} - A_{LP} \times \Delta\theta^o & (IC_{HP/LP}) \\ R_{HP} \geq R_{HA} - A_{HA} \times \Delta\theta^s & (IC_{HP/HA}) \end{array}$$

Consecutively adding $(IC_{LA/LP})$, $(IC_{LP/HA})$, $(IC_{HA/HP})$ assuming $R_{HP} = 0$ and saturating $(IC_{LA/LP})$, $(IC_{LP/HA})$ and $(IC_{HA/HP})$ incentive constraints allows defining the following information rents:

$$R_{LA} = A_{LP} \times \Delta\theta^s + A_{HA} \times (\Delta\theta^o - \Delta\theta^s) + A_{HP} \times \Delta\theta^s \quad (A20) \quad R_{HA} = A_{HP} \times \Delta\theta^s \quad (A22)$$

$$R_{LP} = A_{HA} \times (\Delta\theta^o - \Delta\theta^s) + A_{HP} \times \Delta\theta^s \quad (A21) \quad R_{HP} = 0 \quad (A23)$$

8.3 Second best allocations under the benchmark model

Proof - Lemma 4:

The existence of a separating equilibrium, such as $A_{LP}^{SB} > A_{HA}^{SB}$ is ensured, iff:

$$V_A(A_{LP}^{SB}, \theta_P^s) < V_A(A_{HA}^{SB}, \theta_A^s) \quad (A30)$$

$$\iff \theta_L^o + \theta_P^s + \frac{\mu\Delta\theta^s}{(1-\mu)} < \theta_H^o + \theta_A^s + \frac{\nu(\Delta\theta^o - \Delta\theta^s)}{(1-\nu)\mu} \quad (A31)$$

$$\iff \frac{\mu\Delta\theta^s}{(1-\mu)} < (\Delta\theta^o - \Delta\theta^s) \left[1 + \frac{\nu}{(1-\nu)\mu} \right] \quad (A32)$$

If $\Delta\theta^o - \Delta\theta^s > \Delta\theta^s \iff \Delta\theta^o < 2\Delta\theta^s$.

Then:

$$\frac{\mu}{(1-\mu)} \times \Delta\theta^s < \frac{\mu}{(1-\mu)} \times (\Delta\theta^o - \Delta\theta^s) \quad (A33)$$

Then we write :

$$\frac{\mu}{(1-\mu)} \times \Delta\theta^s < \frac{\mu}{(1-\mu)} \times (\Delta\theta^o - \Delta\theta^s) < (\Delta\theta^o - \Delta\theta^s) \left[1 + \frac{\nu}{(1-\nu)\mu} \right] \quad (A34)$$

Here $\frac{\mu}{(1-\mu)} < 1 + \frac{\nu}{(1-\nu)\mu}$ is a sufficient condition for : $A_{LP}^{SB} > A_{HA}^{SB}$.

Under the assumption $\Delta\theta^o - \Delta\theta^s > \Delta\theta^s$, $\frac{\mu}{(1-\mu)} > 1 + \frac{\nu}{(1-\nu)\mu}$ is a necessary condition for the emergence of a bunching equilibrium between types LP and HA . It can be easily shown that such a bunching equilibrium is less likely to emerge when $\mu < 0.5$. Moreover, $\mu > \nu$ is always verified when such a bunching equilibrium emerges.

Under the alternative assumption (i.e., $\Delta\theta^o - \Delta\theta^s < \Delta\theta^s$), $\frac{\mu}{(1-\mu)} < 1 + \frac{\nu}{(1-\nu)\mu}$ becomes a necessary condition for the emergence of a separating equilibrium and the violation of the latter inequality becomes a sufficient condition for the existence of a bunching equilibrium.

Based on similar calculations, we show that $\frac{\nu}{(1-\nu)\mu} < 1 + \frac{\nu+(1-\nu)\mu}{(1-\nu)(1-\mu)}$ is a necessary condition for the emergence of a separating equilibrium such as: $A_{HA}^{SB} > A_{HP}^{SB}$.

Under the assumption $\Delta\theta^o - \Delta\theta^s > \Delta\theta^s$, $\frac{\nu}{(1-\nu)\mu} > 1 + \frac{\nu+(1-\nu)\mu}{(1-\nu)(1-\mu)}$ is a sufficient condition for the emergence of a bunching equilibrium between the types HA and HP .

Under the alternative assumption (i.e., $\Delta\theta^o - \Delta\theta^s < \Delta\theta^s$), $\frac{\nu}{(1-\nu)\mu} > 1 + \frac{\nu+(1-\nu)\mu}{(1-\nu)(1-\mu)}$ becomes a sufficient condition for the emergence of a separating equilibrium $A_{HA}^{SB} > A_{HP}^{SB}$ and the violation of the latter inequality becomes a necessary condition for the existence

of a bunching equilibrium.

8.4 Second best allocations under a common-value model

Definition of necessary and sufficient conditions for the emergence of separating equilibria under a common-value model

We consider the following functional form:

$$V(A(\theta^{os}), \theta^s) = \alpha \times A^\beta \times (\theta^s)^\gamma$$

Where $\alpha > 1$, $0 < \beta < 1$, $0 < \gamma$ and $\alpha \times \beta \times \gamma \times A^{\beta-1}(\theta^s)^{\gamma-1} > 1$

A separating equilibrium emerges between LA and LP types iff:

$$A_{LP}^{SB} < A_{LA}^{SB}$$

According to first order conditions under asymmetric information we have:

$$V_A(A_{LP}^{SB}, \theta_P^s) = \alpha \times \beta \times (A_{LP}^{SB})^{\beta-1} (\theta_P^s)^\gamma = \theta_L^o + \theta_P^s + \Delta\theta^s \left[\frac{\mu}{1-\mu} \right] \quad (A410)$$

$$\implies A_{LP}^{SB} = \left[\frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)\alpha\beta(\theta_P^s)^\gamma} \right]^{\frac{1}{\beta-1}} \quad (A411)$$

Likewise, according to first order conditions we have:

$$V_A(A_{LA}^{SB}, \theta_A^s) = \alpha \times \beta \times (A_{LA}^{SB})^{\beta-1} (\theta_A^s)^\gamma = \theta_L^o + \theta_A^s \quad (A412)$$

$$\implies A_{LA}^{SB} = \left[\frac{\theta_L^o + \theta_A^s}{\alpha\beta(\theta_A^s)^\gamma} \right]^{\frac{1}{\beta-1}} \quad (A413)$$

Thus a separating equilibrium emerges iff:

$$A_{LP}^{SB} < A_{LA}^{SB} \\ \iff \left[\frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)\alpha\beta(\theta_P^s)^\gamma} \right]^{\frac{1}{\beta-1}} < \left[\frac{\theta_L^o + \theta_A^s}{\alpha\beta(\theta_A^s)^\gamma} \right]^{\frac{1}{\beta-1}}, \text{ yet } \beta - 1 < 0 \quad (A414)$$

$$\iff \frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)\alpha\beta(\theta_P^s)^\gamma} > \frac{\theta_L^o + \theta_A^s}{\alpha\beta(\theta_A^s)^\gamma} \quad (A415)$$

Yet, $\theta_P^s > \theta_A^s \iff (\theta_P^s)^\gamma > (\theta_A^s)^\gamma, \forall \gamma > 0$

$$\iff \frac{\theta_L^o + \theta_A^s}{\alpha\beta(\theta_A^s)^\gamma} > \frac{\theta_L^o + \theta_P^s}{\alpha\beta(\theta_P^s)^\gamma} \quad (A416)$$

Thus the following condition is a necessary one for the existence of a separating equilibrium between LA and LP types:

$$\frac{\theta_L^o + \theta_A^s}{(\theta_P^s)^\gamma} < \frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)(\theta_P^s)^\gamma} \quad (A417)$$

$$\iff 0 < \Delta\theta^s \left[1 + \frac{\mu}{1-\mu} \right] \quad (A418)$$

Based on similar calculations we show:

- $0 < \Delta\theta^s \left[1 + \frac{\mu}{1-\mu} \right]$ (A419) is a necessary condition for the emergence of a separating equilibrium between LA and LP types under asymmetric information for the CV model.

- $0 < \Delta\theta^o + \Delta\theta^s \left[1 + \frac{\nu+(1-\nu)\mu}{(1-\nu)(1-\mu)}\right]$ (A420) is a necessary condition for the emergence of a separating equilibrium between *LA* and *HP* types.
- $0 < (\Delta\theta^o - \Delta\theta^s) \left(1 + \frac{\nu}{(1-\nu)\mu}\right) - \frac{\mu\Delta\theta^s}{1-\mu}$ (A421) is a sufficient condition for the emergence of a separating equilibrium between *LP* and *HA* types.
- $0 < \Delta\theta^s \left[1 + \frac{\nu+(1-\nu)\mu}{(1-\nu)(1-\mu)}\right] - [\Delta\theta^o - \Delta\theta^s] \frac{\nu}{(1-\nu)\mu}$ (A422) is a necessary condition for the emergence of a separating equilibrium between *HA* and *HP* types.

Proof - Proposition 3:

Similarly to Appendix 8.4, we consider first order conditions for the *common-value* model under asymmetric information:

$$V_A(A_{LP}^{SB}, \theta_P^s) = \alpha \times \beta \times (A_{LP}^{SB})^{\beta-1} (\theta_P^s)^\gamma = \theta_L^o + \theta_P^s + \Delta\theta^s \left[\frac{\mu}{1-\mu}\right] \quad (A423)$$

$$\implies A_{LP}^{SB} = \left[\frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)\alpha\beta(\theta_P^s)^\gamma} \right]^{\frac{1}{\beta-1}} \quad (A424)$$

And:

$$V_A(A_{LA}^{SB}, \theta_A) = \alpha \times \beta \times (A_{LA}^{SB})^{\beta-1} (\theta_A)^\gamma = \theta_L + \theta_A \quad (A425)$$

$$\implies A_{LA}^{SB} = \left[\frac{\theta_L + \theta_A}{\alpha\beta(\theta_A)^\gamma} \right]^{\frac{1}{\beta-1}} \quad (A426)$$

Thus a separating equilibrium emerges iff:

$$A_{LP}^{SB} < A_{LA}^{SB} \\ \iff \frac{\theta_L^o + \theta_P^s}{\alpha\beta(\theta_P^s)^\gamma} < \frac{(1-\mu)(\theta_L^o + \theta_P^s) + \Delta\theta^s \mu}{(1-\mu)\alpha\beta(\theta_P^s)^\gamma} \quad (A427)$$

Re-arranging the equation above gives equation (23) (see Section 4.2):

$$A_{LA}^{SB} > A_{LP}^{SB} \text{ iff } \left(\frac{\theta_P^s}{\theta_A^s}\right)^\gamma < \frac{(1-\mu)(\theta_L^o + \theta_P^s) + \mu\Delta\theta^s}{(1-\mu)(\theta_L^o + \theta_A^s)}$$

Equations (24), (25) and (26) are obtained thanks to similar calculations.

8.5 Additionality of joint provision of effective species-specific protection and habitat suitability services (γ)

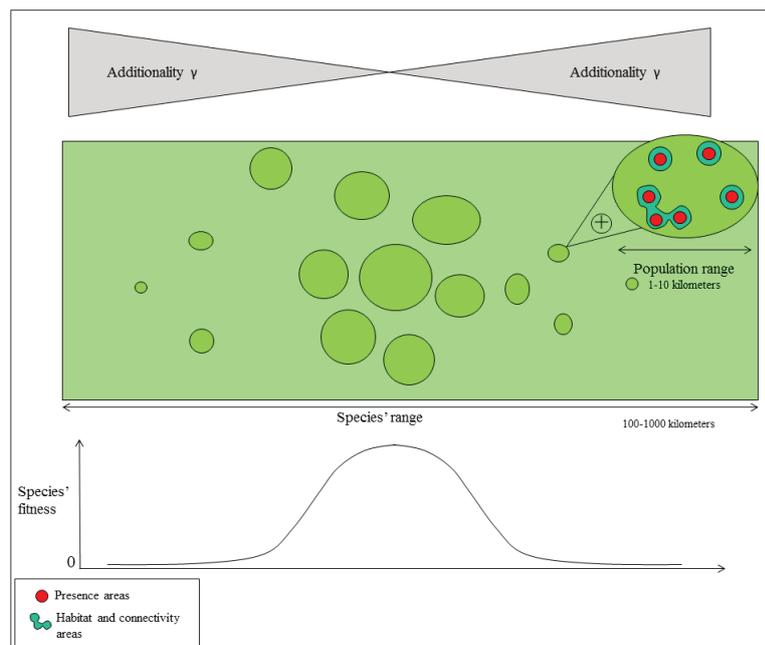
Figure II.8 illustrates a possible definition and interpretation of γ values in a context of species-specific conservation contracts across the whole target species' range. In the core of the species range, environmental conditions are at best for species populations. The provision of sufficient and suitable habitat can suffice to ensure the establishment of ecological networks and species' persistence in this core zone. The conservation planner does not need to focus on effective species populations' presence and specifically target conservation contracts towards properties actually hosting species individuals. The level

of additionality of joined provision of effective species-specific protection and habitat suitability services when compared to the provision of habitat suitability alone is thus in this part of the target species' range.

However, the level of additionality γ increases when dealing with species-specific conservation towards edges of the species' range. Indeed, when designing conservation contracts at the edge of the species' range, the conservation planner aims at maintaining existing species populations experiencing extreme environmental conditions. In this case, the provision of suitable habitat is crucial for the establishment of ecological networks but the latter is useless if it occurs too far away from existing populations. In this part of the species' range, the conservation planner has to uppermost target conservation payments towards properties effectively hosting species populations but also offer payments to neighboring landowners, whose property does not host species populations in order to establish ecological corridors.

As mentioned above, we propose here one example of interpretation of γ values in a static context. However, our model could be applied in a dynamic context in order to – for instance – explain differences in conservation strategies applied when dealing with species-specific conservation contracts aiming at allowing successful climate-forced dispersal under climate change.

Figure II.8: Conservation contracts across target-species' range - Additionality γ



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Chapter III

Cost-effectiveness of conservation payment schemes under climate change

Emeline Hily , Martin Drechsler and Frank Wätzold

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Abstract

Climate change is expected to be one of the key threats for biodiversity conservation in this century. Conservation literature has pointed to the inadequacy of current biodiversity conservation practices relying predominantly on static approaches and showed the need to develop “climate-proof” conservation strategies. However, this debate has taken place largely in the conservation planning literature so far and ignored incentive-based conservation policy instruments such as conservation payments. Our general understanding is thus poor about how should conservation payments be designed so that they can contribute to biodiversity conservation under climate change in a cost-effective manner. In this work we develop an ecological-economic model and investigate the cost-effectiveness of various payment design options involving varying degrees of payments’ differentiation and targeting in a landscape whose dynamics is driven by climate change, while considering the impact of changes in key economic and ecological parameters. We provide the first comparative cost-effectiveness analysis of conservation payment designs in a changing climate on a conceptual level. Our results demonstrate the significant cost-effectiveness gains enabled by payments’ differentiation and targeting for biodiversity conservation under climate change. Moreover, we demonstrate the existence of connectivity/area trade-offs under climate change. The cost-effectiveness performance of targeted payments compared to untargeted differentiated payments increase with decreasing species dispersal ability but decrease with decreasing climate stability in the landscape.

Keywords: Biodiversity, Conservation payments, Cost-effectiveness, Climate change, Ecological-economic modeling, Spatio-temporal dynamics

1 Introduction

Climate change is expected to be one of the key threats for biodiversity conservation in this century (Sala *et al.*, 2000). Thomas *et al.* (2004) estimated extinction risks due to climate change for selected regions that cover approximately 20% of the Earth's terrestrial surface. Their results suggest that 15–37% of species will be committed to extinction by 2050. Besides, numerous studies have evidenced species range shifts caused by climate change (Parmesan and Yohe, 2003; Julliard *et al.*, 2004, Tamis *et al.*, 2005). On that basis, many authors have pointed to the inadequacy of current biodiversity conservation practices relying predominantly on static approaches such as spatially fixed areas for conservation (Hannah *et al.*, 2007; Hannah, 2010). Instead, they argued for the need to develop “climate-proof” conservation strategies that either provide the species with the opportunity to disperse to “new climate space” or enhance the habitat quality in a way that enables the species to survive in the present range (Vos *et al.*, 2008). However, this debate has taken place largely in the conservation planning literature (Vos *et al.*, 2008, Hodgson *et al.*, 2009, Oliver *et al.*, 2016) and ignored incentive-based conservation policy instruments such as payments for landowners to carry out biodiversity-enhancing land use measures (henceforth referred to as conservation payments).

Notable exceptions exist. Ay *et al.*, (2014) investigated effects of climate change and land use change on birds' abundances in France, and showed that a uniform agri-environment payment to promote pastures would counteract negative impact of climate-induced land-use change on common bird abundancies. Tainio *et al.*, (2016) compared the cost-effectiveness of different agri-environment schemes with other conservation policies such as translocation and dispersal corridors to conserve butterflies under climate change in Finland, and found that supporting buffer zones by agri-environment schemes is a cost-effective measure. However, our general understanding is poor about how should conservation payments be designed so that they can contribute to biodiversity conservation under climate change in a cost-effective manner.

Designing conservation payments in a region with a shifting area of potentially suitable habitat is not straightforward. Consider a case in which, initially, conservation measures generate suitable habitats only in the southern part of a region. Then, due to climate change the area of potentially suitable habitat moves northwards and, finally,

conservation measures can generate suitable habitats only in the northern part of the region. A real world example for such a scenario are specific mowing or grazing regimes to conserve European grassland butterflies whose suitable habitat range shifts northwards. The mowing regimes are only effectively conserving the species if they are carried out in the area of potentially suitable habitat (O'Connor *et al.*, 2014, Johst *et al.*, 2015).

Now consider *uniform payments* that all landowners receive who carry out a conservation measure in a region. Such payment type is frequently applied to conserve species, for example in the context of many European agri-environment schemes (Armsworth *et al.*, 2012; Wätzold *et al.*, 2016). However, in a situation with a shifting habitat suitability their drawback is that all landowners in a region receive a payment for conservation measures irrespective of whether their land is in an area with suitable climate, i.e. irrespective of whether the conservation measure is effective.

A possible alternative design option are *regionally differentiated payments* that consider heterogeneity in terms of climate-suitability for the conserved biodiversity within a region. The idea of such a design option is that payments are differentiated across sub-regions depending on the climate-suitability of the sub-region. However, with climate change, the differentiation across sub-regions has to change as well in order to reflect shifts of the area of suitable habitat. In contrast to uniform payments, regionally differentiated payments induce conservation measures only on land which is a potentially suitable habitat. The disadvantage in comparison to uniform payments is that transaction costs are substantially higher. Generally, regionally differentiated schemes are more costly to administer for the conservation agency (Armsworth *et al.*, 2012; Hanley *et al.*, 2012). Moreover, the conservation agency needs reliable information about climate change in the region and how climate change affects the biodiversity to be conserved. If a species to be conserved is not present throughout the area with suitable habitat, conservation measures may also be carried out on land where the species is absent and which cannot be reached by the species.

A third possible payment design are payments targeted at the presence of a species in an area (*targeted uniform payments*). Here, the conservation agency addresses shifting habitat suitability and species range by paying only landowners for conservation measures whose land is either occupied by a species or located near an occupied area. The design feature that land around an occupied area can be conserved, allows the species to

move in a landscape by slowly occupying conserved land which is near already occupied land and in new climate space. Thus, the eligibility of land for conservation measure moves in line with the habitat suitability. If land is not suited anymore as a habitat for a species due to climate change, the species will not be present anymore. Therefore, the land will not be any more eligible for conservation measures. Unlike in the uniform payment scheme, money is not wasted on financing ineffective conservation measures on land which provides, in principle, suitable habitat but on which the species is not present and which can also not be reached by the species. Moreover, for targeted uniform payments, the conservation agency does not need detailed knowledge about regional climate change and its effect on species. However, transaction costs are certainly higher for the administration of the targeted uniform scheme than for the uniform scheme. In particular, they are likely to be high for monitoring the species' presence and absence all over the region. Additionally, the same mechanism that allows species to move in the landscape — not only land where the species is present but also neighbouring land is eligible for conservation measures — also generates some ineffectiveness. Conservation measures may be carried out on land near the southern border of the area of suitable habitat, which is still close to land occupied by the species but not suitable anymore for conservation due to climate change.

This ineffectiveness can be avoided with a fourth payment design option that combines the ideas of a regionally differentiated payment scheme and a targeted scheme (*targeted regionally differentiated scheme*). Here, landowners only receive a payment for a conservation measure if they fulfil two conditions. The conservation measure is carried out on land that is potentially suitable as a habitat and, the species to be conserved is present on this land or on an area close to this land. Because of these two conditions, payments are targeted more effectively at suitable land for conservation than with the other three payment schemes. However, transaction costs are also likely to be higher than for the three other schemes. As with the regionally differentiated scheme, the conservation agency needs reliable information about climate change and how its change affects the biodiversity to be conserved. Additionally, it has to monitor the absence and presence of the species in the area of suitable habitat.

In summary, the four conservation payment design options face different trade-offs in terms of the effective selection of land suitable for conservation measures and transaction

costs. The aim of this paper is to improve our understanding of the cost-effectiveness of the different options (understood as maximizing species conservation for given conservation budgets) in a landscape whose dynamics is driven by climate change. We investigate, with a conceptual ecological-economic model, the relative cost-effectiveness performance of the four design options depending on changes in economic, ecological and climate parameters defining the conservation context.

Our work builds on previous research about the cost-effectiveness of conservation payments. Integrated ecological-economic modelling has been a successful method for this type of analysis (Barraquand and Martinet, 2011; Bauer and Swallow, 2013; Mouysset *et al.*, 2015). Similar to Lewis *et al.*, (2009), Drechsler *et al.*, (2010) and Wätzold and Drechsler (2014), we apply a conceptual model to analyze the cost-effectiveness of conservation payments. Regarding payment design, we are in the tradition of Babcock *et al.*, (1997), Ferraro (2003) and Duke *et al.*, (2014) who analyze the cost-effectiveness of targeted payments and Lewis *et al.*, (2011), Armsworth *et al.*, (2012) and Wätzold *et al.*, (2016) who analyze the cost-effectiveness of regionally differentiated payments. However, none of this research has considered climate change. In contrast, Leroux and Whitten (2014) and Mallory and Ando (2014) considered the impact of climate change on conservation policies from an economic perspective. However, they addressed the ranking of competing conservation projects (Leroux and Whitten, 2014) and the prioritization of conservation areas (Mallory and Ando, 2014) but did not investigate the policy instrument of conservation payments.

Our work is novel in two ways. The first novelty refers to the area of application. We provide the first cost-effectiveness analysis of different conservation payment designs in a changing climate on a conceptual level. The second novelty refers to the modelling approach. Previous conceptual ecological-economic models to analyze conservation payments (Lewis *et al.*, 2009, Drechsler *et al.*, 2010, Wätzold and Drechsler, 2014) are static. We consider a dynamic landscape with climate change-induced habitat shifts and model feedback loops between the conservation measures incentivized by payments and the presence of species. We need to consider feedback loops to analyze the cost-effectiveness of the targeted conservation payments.

2 The Model

2.1 The conservation problem

A conservation agency aims at the conservation of a target species in a heterogeneous landscape under climate change by means of incentive payments. We consider a C columns \times J rows landscape divided into N individual land parcels corresponding to N individual properties. For simplicity, we assume that each individual land parcel $i \in \llbracket 1; N \rrbracket$ belongs to a single individual owner. At each time period, a landowner decides either to use the land parcel for economic production, such as agriculture or forestry, or to conserve it. When participating in the conservation program, the landowner gives up on income derived from economic production and thus incurs opportunity costs (henceforth referred to as conservation costs).

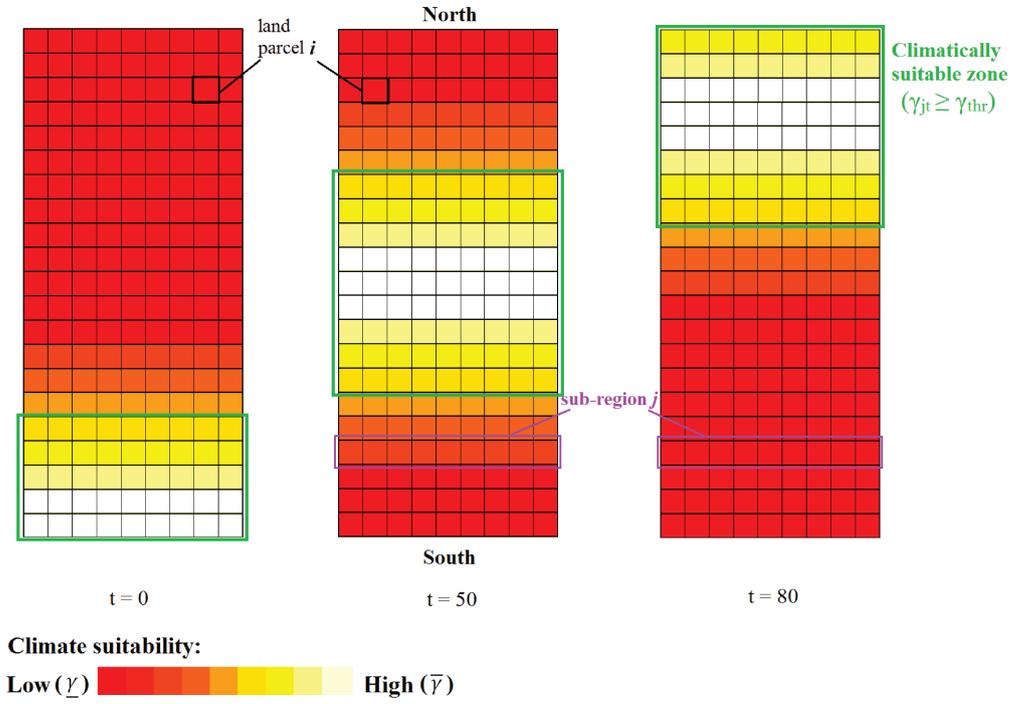
Climate-suitability — according to the target species' niche — varies across the landscape and the J rows, henceforth referred as sub-regions, show heterogeneous levels of climate-suitability. Let γ_{jt} designate the level of climate-suitability in the sub-region $j \in \llbracket 1; J \rrbracket$ at time t , with $\gamma_{jt} \in [\underline{\gamma}, \bar{\gamma}]$. We consider that all land parcels located in the same region j show the same level of climate suitability γ_{jt} . Moreover, the target species undergoes a climate-forced range shift due to climate change. Therefore, climate-suitability characteristics of sub-regions (and thus of land parcels) also vary over time. Figure III.1 displays the landscape's structure as well as the variation of climate-suitability γ_{jt} over space and time in the landscape. In Figure III.1, the climatically suitable area is delimited by the green rectangle and corresponds to all cells (i.e., land parcels) whose level of climate suitability γ_{jt} exceeds a threshold value γ_{thr} .

We assume that the conservation agency has *a priori* no information about the exact level of climate-suitability in the sub-regions of the landscape for the target species. However, the agency can gather information about regional climate-suitability levels γ_{jt} in the whole landscape; let T designate the level of transaction costs borne by the conservation agency per time period t in this case.

The conservation agency has no information about the location of target species' populations in the landscape and has to perform a costly monitoring in order to determine whether a land parcel is occupied. Let m designate the monitoring cost for one

land parcel.

Figure III.1: Variation of climate-suitability over space and time in the landscape: The subregions of high climate suitability move from south to north.



Finally, land parcels show heterogeneous conservation costs characteristics. The individual conservation cost c_i of a land parcel i ranges uniformly between $\bar{c} - \sigma$ and $\bar{c} + \sigma$ in the landscape and each c_i is known only to landowner i . We assume that the conservation agency only has information on the conservation costs' distribution in the interval $[\bar{c} - \sigma, \bar{c} + \sigma]$ and the landowners' decision rule for participation. A landowner participates in the program if and only if the conservation payment is higher than the conservation cost.

In this context, the objective of the conservation agency is to ensure a successful climate-forced dispersal and maximize species' survival. To do this, the conservation agency considers four alternative designs of conservation payments schemes aiming at inducing the implementation of conservation measures by landowners in the landscape:

- A uniform payment (U)
- A regionally differentiated payment (RD)
- A targeted uniform payment (TU)

- A targeted regionally differentiated payment (TRD)

3 Incentive conservation payments

3.1 Uniform payment (U)

In the case of a uniform payment the conservation agency does not account for regional heterogeneity in terms of climate-suitability and offers a uniform payment P_t at each time period t to landowners, based on the level of available conservation budget B_t at time t and conservation costs' distribution. The level of payment P_t is defined so as to exhaust the available budget B_t , while accounting for landowners' decision rule for participation x_{it} (x_{it} equals 1 when property i is conserved at time period t , 0 otherwise). Besides, the definition of the level of payment is subject to the following constraint: $\bar{c} - \sigma \leq P_t \leq \bar{c} + \sigma$. The level of payment P_t is thus defined in the following way:

$$\left\{ \begin{array}{l} \sum_{i=1}^N x_{it} \times P_t = B_t \\ \text{s.t. } \bar{c} - \sigma \leq P_t \leq \bar{c} + \sigma \end{array} \right. \quad (\text{III.1})$$

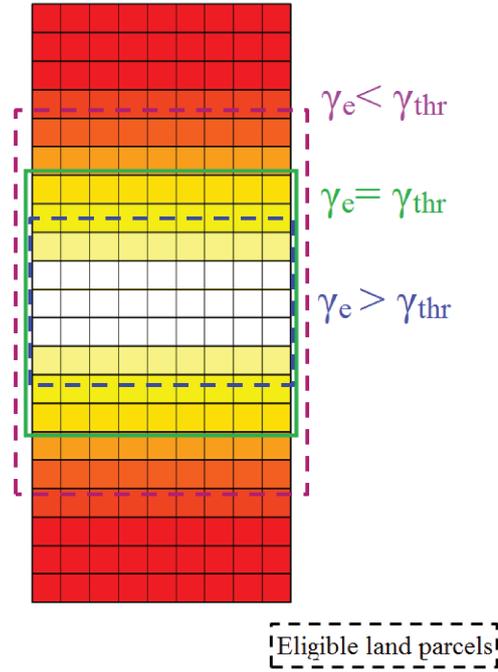
3.2 Regionally differentiated payment (RD)

In this case, the conservation agency accounts for the heterogeneity in terms of regional climate-suitability γ_{jt} through regionally differentiated payments. For the differentiation, the conservation agency defines a threshold value γ_e for climate suitability determining eligibility for the payment. For each time period t , the agency offers a payment P_{jt} only to owners whose land parcels are located in sub-regions showing a level of climate suitability greater than γ_e (see Figure III.2). Thus, let ϵ_{jt} define the eligibility of region j for the payment at time t :

$$\left\{ \begin{array}{l} \epsilon_{jt} = 1 \quad \text{if } \gamma_{jt} \geq \gamma_e \\ \epsilon_{jt} = 0 \quad \text{otherwise} \end{array} \right. \quad (\text{III.2})$$

The conservation agency exhausts the available conservation budget B_t and defines the payment P_{jt} while accounting for landowners' decision rule for participation x_{it} as

Figure III.2: Eligibility rule under regionally differentiated payments (RD)



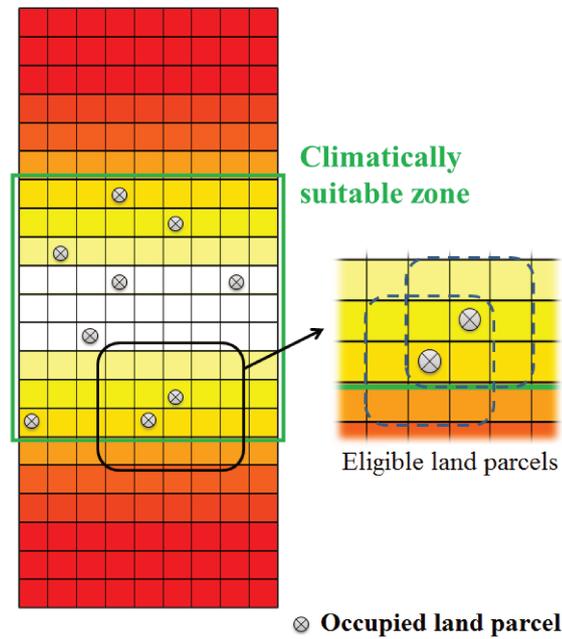
well as for disparities in terms of regional climate-suitability γ_{jt} through the eligibility rule ϵ_{jt} . As mentioned earlier in Section 2.1, gathering information about the exact level of climate-suitability in the J sub-regions of the landscape is necessary prior to payment's definition. Consequently, the conservation agency incurs transaction costs T at each time period t . The level of payment P_{jt} is thus homogeneous for all sub-regions j being eligible for the payment at time t . Besides, eligibility and thus also payment vary over time for a sub-region j . The level of payment P_{jt} is here defined in the following way:

$$\begin{cases} \sum_{j=1}^J \epsilon_{jt} \sum_{i=1}^{N_j} x_{it} \times P_{jt} + T = B_t \\ s.t. \quad \bar{c} - \sigma \leq P_{jt} \leq \bar{c} + \sigma \end{cases} \quad (\text{III.3})$$

As mentioned in section 3.1, the level of payment P_{jt} cannot be higher than the maximum level of conservation cost $\bar{c} + \sigma$. In the case of RD payments, the stringency of the eligibility rule combined with this constraint can lead to the non-exhaustion of the conservation budget for high levels of budget. This phenomenon is observed for TU and TRD payments as well. We consider that the leftover budget is lost and is not

transferred to the next time period.

Figure III.3: Eligibility rule under targeted uniform payments (TU)



3.3 Targeted uniform payment (TU)

In the case of TU payments, only landowners whose property is either occupied by a species' population or located in the direct neighbourhood of an occupied land parcel are eligible to the scheme (see Figure III.3). As mentioned earlier, the conservation agency has no information about the location of target species' populations in the landscape and a preliminary monitoring is necessary for each land parcel of the landscape in order to identify eligible land parcels. Let m designate the monitoring cost for one property. At each time period t , the agency offers a payment P_t which is uniform across the landscape. Let S_{it} designate a random variable equalling 1 if the central land parcel i of a neighbourhood is occupied and 0 otherwise. Finally, n_{-i} designates the number of properties in a direct neighbourhood surrounding property i (i.e., 8). The conservation agency defines the level of payment P_t so as to exhaust the available conservation budget B_t , while accounting for landowners' decision rule for participation x_{kt} in a neighbourhood, eligibility rule S_{it} and monitoring costs m . The level of payment

P_t is here defined in the following way:¹

$$\left\{ \begin{array}{l} \sum_{i=1}^N (m + S_{it} \sum_{k=1}^{n-i+1} x_{kt} \times P_t) = B_t \\ s.t. \quad \bar{c} - \sigma \leq P_t \leq \bar{c} + \sigma \end{array} \right. \quad (\text{III.4})$$

3.4 Targeted regionally differentiated payment (TRD)

In this case, the conservation agency not only targets the payment towards landowners whose property is either occupied or located in the direct neighbourhood of an occupied land parcel but also accounts for disparities in terms of regional climate-suitability γ_{jt} in the payments' definition (see Figure III.4). In this case the conservation agency defines the level of payment P_{jt} at each time period t so as to exhaust the available conservation budget B_t , while accounting for landowners' decision rule for participation x_{kt} in a neighbourhood, eligibility rule S_{it} , disparities in terms of regional climate-suitability γ_{jt} through the eligibility rule ϵ_{jt} as well as for monitoring (m) and transaction costs (T). In this case and contrary to targeted uniform payments, only climatically suitable land parcels are monitored in order to detect the presence of species' individuals. The level of payment P_{jt} is here defined in the following way:

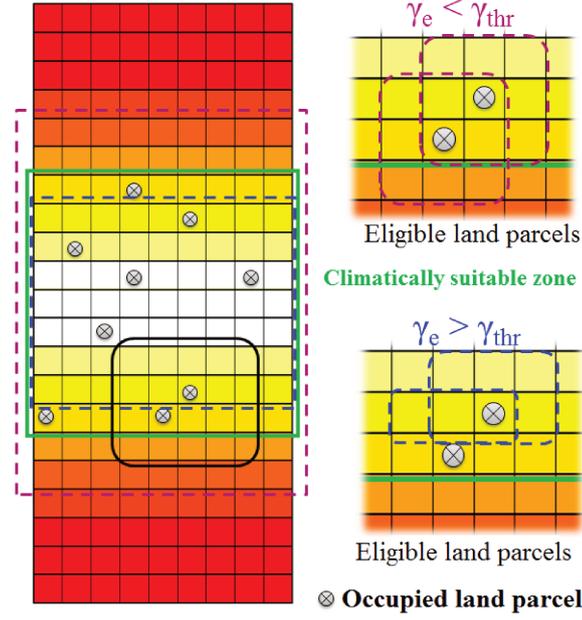
$$\left\{ \begin{array}{l} \sum_{j=1}^J \epsilon_{jt} \sum_{i=1}^{N_j} (m + S_{it} \sum_{k=1}^{n-i+1} x_{kt} \times P_{jt}) + T = B_t \\ s.t. \quad \bar{c} - \sigma \leq P_{jt} \leq \bar{c} + \sigma \end{array} \right. \quad (\text{III.5})$$

3.5 The ecological benefit

We determine the ecological output of considered conservation payment schemes by using the Hanski's (1999) metapopulation model which describes the dynamics and survival of species in fragmented landscapes. The species dynamics are characterised by local extinction of local populations and recolonization of empty land parcels by neighbouring populations.

¹ When simulating the implementation of targeted uniform payments (TU) and targeted regionally differentiated payments (TRD) each land parcel can be conserved only once at each time period, even if the latter belongs to several direct neighbourhoods of an occupied land parcel.

Figure III.4: Eligibility rule under targeted regionally differentiated payment (TRD)



Survival and colonization can be successful in suitable land parcels only. A land parcel i is considered suitable if it is climatically suitable for a species and conserved at time t . As mentioned earlier, a land parcel is climatically suitable if the level of climate suitability γ_{jt} in the considered land parcel is higher than a certain threshold value γ_{thr} . A local population goes extinct in an occupied land parcel i at rate e_{it} if the land parcel is suitable at time t . If the land parcel becomes unsuitable (due to climate change) the local population goes extinct with certainty. Finally, an empty land parcel i is colonized at rate τ_{it} if the land parcel is suitable at time t as shown in Equation (III.6) below. Here, Im_{it} corresponds to the immigration rate (i.e., number of immigrants per time unit) in a land parcel i and θ designates the minimum number of immigrants required for a successful colonization.

$$\begin{cases} \tau_{it} = \frac{Im_{it}^2}{Im_{it}^2 + \theta^2} & \text{if } \gamma_{jt} \geq \gamma_{thr} \text{ and } x_{it} = 1 \\ \tau_{it} = 0 & \text{otherwise} \end{cases} \quad (\text{III.6})$$

The immigration rate into land parcel i is given by:

$$Im_{it} = \sum_{k=1}^K \pi_{kt} \nu \frac{\exp(-d_{ik}/d)}{S_t} \quad (\text{III.7})$$

Here, π_{kt} equals 1 if the land parcel k located at a distance d_{ik} from the central land parcel i is conserved and occupied by the species, π_{kt} equals 0 otherwise. ν designates the emigration rate from land parcel k (i.e., the number of individuals emigrating from land parcel k per time period). S_t corresponds to the number of suitable land parcels in the neighbourhood of the central land parcel i and d designates the target species mean dispersal distance.

To assess the ecological benefit of each payment scheme, we simulate for each payment design option the land-use dynamics (pattern of suitable — i.e., climatically suitable and conserved — land parcels), and on the basis of these dynamics the species dynamics is simulated according to the model described above. Additionally, in the case of targeted uniform payments and targeted regionally differentiated payments there is a feedback loop between the land-use dynamics and the species dynamics because the level of conservation payment for a given land parcel depends on the species occupancy in the neighbourhood.

For each payment design option we determine the mean time to species extinction over numerous simulation replicates as well as the probability of surviving the time frame of the simulation. Both quantities will be used as indicators of species viability for a chosen payment design option. We base our analysis on these two criteria as they are complementary for the assessment of the level of ecological benefit derived from each payment design option. Indeed, measuring the mean time to species extinction in the landscape does not allow to detect whether the implementation of a payment scheme has allowed the species to survive the climate-forced range shift.

In addition to these indicators we measure — for each payment design option — the average number of conserved land parcels as well as the number of climatically suitable conserved land parcels. We also measure the average number of conserved direct neighbours around a conserved land parcel (which measures whether the conserved land parcels are clustered or dispersed in the study region). Finally, we measure the average number of climatically suitable conserved direct neighbours per climatically suitable conserved land parcels, which allows to measure the degree of clustering of climatically

suitable conserved land parcels in the landscape.

4 Analysis

4.1 Numerical simulations at the landscape scale

We investigate the relative cost-effectiveness performances of the four conservation payment schemes under climate change by numerically simulating the implementation of the schemes at the landscape scale. In order to determine the impact of key economic and ecological parameters on payment schemes' relative cost-effectiveness we simulate the implementation of each payment design option with different values for these key parameters. We study the impact of variations in the level of the conservation budget B_t , the degree of conservation cost heterogeneity in the landscape σ , the magnitude of transaction cost T and the magnitude of monitoring cost m on payment schemes' relative cost-effectiveness. We also investigate the impact of a variation in the level of stringency of the eligibility rule for differentiated payment schemes (i.e., value of the γ_e parameter), species mean dispersal distance d (i.e., dispersal ability) and the level of climate stability in the landscape. A unique combination of parameter values defines a conservation scenario. Here we present the main features of the numerical simulations as well as the various scenarios considered.

Landscape structure and dynamics

We consider a 10 columns \times 20 rows landscape consisting of $N = 200$ individual land parcels. The individual conservation cost c_i of a land parcel is drawn from a uniform distribution $U(\bar{c} - \sigma, \bar{c} + \sigma)$. We consider that the distance between two direct neighbouring land parcels d_{ik} is equal to 1.

We simulate the climate-forced shift of the target species' range in the following way: climate-suitability γ_{jt} of a row or sub-region j varies from 0 to 1 and is calculated based on Equation (III.8) below (corresponding to a bell-shaped climate-suitability). As mentioned earlier, climate-suitability varies across the landscape: land parcels that are located in the same row show the same level of climate-suitability but the latter varies among rows (see Fig. III.1). Moreover, the mean of the bell-shaped climate-suitability curve is shifted time step by time step towards the northern part of the landscape as

displayed in Figure III.1. An increase in the level climate stability in the landscape leads to a slower species' range shift. The climatic suitability of a land parcel i at time t is calculated as:

$$\gamma_{jt} = \exp\left(\frac{-(j - \mu)^2}{2\sigma^2}\right) \quad (\text{III.8})$$

Here μ corresponds to the coordinate of the central row (maximum climate-suitability) and ρ is the standard deviation of the climate-suitability curve.

As shown earlier, climate-suitability characteristics of a land parcel i and the conservation status of the land parcel x_{it} jointly determine both the colonization rate τ_{it} of this land parcel at time t as well as the extinction rate e_{it} of a population in the land parcel. Equation (III.9) below shows how the extinction probability e_{it} in a land parcel i is related to sub-regional climate suitability γ_{jt} . Extinction rate e_{it} is identical for all *occupied* land parcels located in the sub-region j .

$$\begin{cases} e_{it} = 1 - 0.9 \times \gamma_{jt} & \text{if } \gamma_{jt} \geq \gamma_{thr} \text{ and } x_{it} = 1 \\ e_{it} = 1 & \text{otherwise} \end{cases} \quad (\text{III.9})$$

Finally, for each simulation run we consider a time frame of 100 time periods. Table III.1 hereinafter displays the values chosen for the invariant parameters for the simulations.

Considered scenarios

Parameter values considered for the definition of the different conservation scenarios are presented in Table III.2. We consider 100 repetitions for each conservation scenario, which allows us to account for the randomness of the hypothetical landscape, the land-use dynamics and the species dynamics. We then perform our analysis on averages of the above-mentioned species' viability indicators (i.e., mean time to species extinction and probability of non-extinction).

Table III.1: Parameter values

Parameter	Description	Value
\bar{c}	Mean conservation cost	1
d_{ik}	Distance between direct neighbouring land parcels	1
θ	Number of immigrants required for a successful colonization of an empty land parcel	5
$\nu_k = \nu$	Emigration rate from neighbouring land parcel k	100
γ_{thr}	Climate suitability threshold	0.5
ρ	Standard deviation of the climate-suitability curve	4

Table III.2: Variables

Variable	Description	Considered range
B_t	Level of available conservation budget at time t	{10, 20, 30, 40, 50, 60, 70, 80, 90, 100}
σ	Cost heterogeneity	{0.05, 0.1, 0.15, 0.2, 0.25, 0.3}
d	Species' Mean dispersal distance	{1, 2, 3, 4, 5}
Climate stability	Time frame with stable climate (in time periods)	{1, 2, 3, 4, 5, 6, 7}
m	Monitoring cost (land parcel occupation)	{0.01, 0.05, 0.1}
T	Transaction cost (climate-suitability)	{2, 10, 20}
γ_e	Payment eligibility threshold (climate suitability)	{0.3, 0.5, 0.7}

5 Econometric analysis

5.1 Pairwise comparison of cost-effectiveness performances of the four considered payment schemes

We compare the relative cost-effectiveness of the four considered payments schemes through a statistical analysis. Performing a statistical analysis instead of a graphical analysis allows us to identify the impact of each parameter of interest on the relative

cost-effectiveness of the various considered payment schemes in a more straightforward and compact way. This statistical approach is valid as long as it is made sure that significant trends highlighted by statistical results are coherent from both an economic and ecological point of view and are not related to aberrant or marginal phenomena. These issues can be tested through the examination of box-plots and the identification of aberrant points. As explained earlier, for each observation we assess conservation results derived from the implementation of a specific payment scheme with two criteria: the mean time to species' extinction (henceforth referred to as mean extinction time) in the landscape as well as the mean probability of the species surviving the time frame of the simulation.

Here we consider pairwise comparisons of cost-effectiveness performances for the four considered payments schemes based on the mean extinction time criteria (resp. survival probability). For each pairwise comparison and for each unique conservation scenario we calculate Δ , which corresponds to the difference between the mean extinction time (resp. survival probability) in the landscape obtained under a payment scheme (1) — considered as the reference payment scheme — and another payment scheme (2). We compare the performance of each type of payment scheme to the performance of payment schemes showing the next level of sophistication. This allows us to study the relative cost-effectiveness of the four payment design options by transitivity. We thus consider the following four pairwise comparisons: TU-U, RD-U, TRD-TU and TRD-RD. Besides, we study the relative performance of TU and RD payments. It is particularly interesting to consider this TU-RD comparison since the definition of these two types of payments and especially the associated eligibility rules substantially differ. We thus end up with considering five pairwise comparisons. We study relative cost-effectiveness performances of the two payments schemes through two types of econometric models: a binary logit model and a multiple linear regression model.

Estimating these two types of models allows us to directly compare the relative cost-effectiveness of the four considered payment types as well as the influence of the conservation context on relative cost-effectiveness performances of payment schemes in a complementary manner. Indeed, estimating a logit model allows to identify the determinants — in terms of conservation context — favouring a strictly better conservation

result derived from each payment design option when compared to others. However, as explained above the estimation of a logit model requires summarizing and thus reducing the information about the difference in terms of ecological benefit between the two payment schemes Δ in the form of a dummy variable. Estimating a multiple linear regression model allows to identify the impact of conservation context variables on the evolution of this difference Δ .

For each pairwise comparison, we create a dummy variable indicating for each conservation scenario whether the reference payment scheme has led to a strictly higher mean extinction time (resp. survival probability) in the landscape compared to the second payment scheme (i.e., $\Delta > 0$). We then estimate the probability for the reference payment scheme to allow a strictly better conservation result as a function of the two payment schemes' characteristics and conservation context through a binary logit model. We consider the following econometric model:

$$\text{logit } p_\beta(x) = \beta_0 + \beta_{P1}X_{P1} + \beta_{P2}X_{P2} + \beta_C X_C + \epsilon_L \quad (\text{III.10})$$

Where $p_\beta(x) = P(Y = 1_{/X=x})$ and $\text{logit } p_\beta(x) = \ln \frac{p_\beta(x)}{1-p_\beta(x)}$

Here $Y = 1$ refers to the achievement of a strictly higher mean extinction time (resp. survival probability) under the reference payment type (i.e., $\Delta > 0$); X_{P1} refers to characteristics of the reference payment scheme (e.g., level of monitoring and transaction cost, etc.), X_{P2} refers to characteristics of the second payment scheme considered, X_C corresponds to variables defining the conservation context (e.g., species mean dispersal distance, climate stability, etc.) and ϵ_L corresponds to the error term.

We also estimate the difference between the mean extinction time (resp. survival probability) in the landscape obtained under the reference payment scheme and a second payment scheme as a function of the two payment schemes' characteristics and conservation context through a multiple linear regression model. We consider the following econometric model:

$$\Delta = \alpha_0 + \alpha_{P1}X_{P1} + \alpha_{P2}X_{P2} + \alpha_C X_C + \epsilon_R \quad (\text{III.11})$$

Where ϵ_R corresponds to the error term.

Data: Simulation results

For computational time reasons, it was not possible to consider all possible combinations of parameter values. We consider 600 scenarios for the conservation context (i.e., unique combinations of parameter values for conservation budget, level of cost heterogeneity, species' mean dispersal distance and climate stability variables) and 25 different types of payments (i.e., unique combinations of parameter values for monitoring cost, transaction cost and payment eligibility threshold variables).

For each simulation run, we measure the ecological benefit derived from of a specific payment type while considering the joint variation of two conservation context variables for which we consider all combinations of parameter values. We consider the following joint variations: budget and species' mean dispersal distance, species' mean dispersal distance and climate stability, budget and climate stability and finally budget and level of cost-heterogeneity in the landscape. When studying the impact of each of these four joint variations on payment's performance, we perform simulations for all parameter combinations for the two variables of interest for several parameter values of the other conservation context variables. When studying the impact of a joint variation in the level of conservation budget and species' mean dispersal distance, we consider two values for climate stability: low and high climate stability (i.e., $Clim_stab \in \{1, 5\}$). When studying the impact of a joint variation in the level of conservation budget and climate stability, we consider at least three values for species' mean dispersal distance: $Disp_dist \in \{1, 2, 5\}$. When studying the impact of a joint variation the level of climate stability and species' mean dispersal distance, we consider two values for the budget variable: low and high budget (i.e., $B_t \in \{20, 70\}$). For these simulation runs, there is no variation in the level of conservation cost heterogeneity in the landscape (i.e., σ) and we consider a base-case value $\sigma = 0.1$. We specifically study the impact of a joint variation in the level of conservation cost heterogeneity in the landscape and conservation budget on the various payment schemes' performances while considering three values for species' mean dispersal distance ($Disp_dist \in \{1, 2, 5\}$) and one value for the level of climate stability ($Clim_stab = 5$).

We considered all combinations of parameter values when studying the impact of payment schemes' characteristics (i.e., level of transaction and monitoring cost as well as eligibility threshold) on ecological benefit for TU and RD payments. When studying

performances of TRD payments, we consider all values for the eligibility threshold parameter (i.e., $\gamma_e \in \{0.3, 0.5, 0.7\}$) and the two lower values for monitoring cost and transaction costs parameters (i.e., $m \in \{0.01, 0.05\}$ and $T \in \{2, 10\}$).

Finally, we had to perform additional simulation runs to specifically measure the performance of TU and RD payments regarding the number of land parcels conserved and the degree of clustering of conserved land parcels. For these simulation runs, we consider all combinations of parameter values for species' mean dispersal distance and level of conservation budget variables for at least five values for the climate stability variable (i.e., ($Clim_stab \in \{1, 2, 4, 5, 7\}$)). In this case, simulations are performed while considering the base-case value for the level of cost-heterogeneity in the landscape (i.e., $\sigma = 0.1$).

As mentioned in Section 3.2, for more sophisticated types of payments (i.e., TU, TRD and RD payments) we could observe that for high levels of the available conservation budget, the budget is not exhausted. This is due to the existence of eligibility criteria for these payments (i.e., occupation of the land parcel, climate suitability): only a limited number of land parcels meet the eligibility criteria and for high levels of conservation budget a fraction of the budget is sufficient to conserve them all. In order to avoid distortion of simulation results we consider only observations for which the difference between the level of available conservation budget and the level of spent budget is strictly smaller than the size of the conservation payment. Finally, we perform our analysis on 13,930 observations. Definition of variables and descriptive statistics are presented in Table III.9 (see Appendix 9.1).

6 Results

Estimation results and marginal effects of the binary logit models are displayed in Tables III.3(a) to III.3(e). Results of the multiple linear regression models are displayed in Tables III.4(a) to III.4(e).² Tables III.5 to III.8 display results of the TU-RD comparison in terms of number of land parcels conserved and the degree of clustering of conserved land parcels.

Results generally show (Tables III.3 and III.5 to III.8) that the implementation of more sophisticated payments leads to a better cost-effectiveness performance. Thus, TU and

²The performance of a Breusch-Pagan test reveals that heteroscedasticity occurs for all considered multiple linear regression models. Results displayed are robust to heteroscedasticity.

RD payments generally perform better than U payments, and TRD payments show a better conservation performance compared to RD and TU payments. Therefore, in the following sections we focus on the TU-RD comparison and seek to identify the determinants of a better cost-effectiveness of targeted payments relatively to differentiated payments and vice versa.

Finally, when comparing payment schemes regarding species viability in the landscape, we only display results of pairwise comparisons in terms of mean extinction time since results related to the probability of non-extinction are similar.

6.1 Impact of variations in economic parameters

Level of conservation budget B_t

First, it can be noticed that an increase in the conservation budget has a significant negative impact on the probability for TU and RD payments to achieve a strictly better conservation results compared to U payments, all else being equal (see Tables III.3(a) and III.3(b)). Indeed, the coefficient related to the *Budget* variable shows a negative sign and is significantly different from zero in both models. This result seems to point to the significant cost-effectiveness gains by implementing such schemes compared to U payments. Nevertheless, Tables III.3(a) and III.3(b) show that the marginal effect associated to the *Budget* variable is low in both cases. Furthermore, results of the linear regression model (Table III.4(a)) show that the difference between the mean extinction time obtained under RD payments and obtained under U payments significantly increases with an increase in the available conservation budget. A similar result is obtained for the TU-U comparison (see Table III.4(b)). Considering the results of the linear regression models it can be observed that values of coefficients associated to the budget variable are also low (Tables III.4(a) and III.4(b)). Since results obtained with multiple linear regression and binary logit models do not concur, we cannot unambiguously conclude about the better cost-effectiveness of TU and RD payments compared to U payments under climate change.

However when considering the RD-TU comparison, results obtained from the logit model and the regression model concur (see Tables III.3(c) and III.4(c)). RD payments seem to be slightly more cost-effective than TU payments and RD payments achieve

strictly better conservation results under low and intermediate levels of conservation budget compared to TU payments. This result can be explained by the difference, in terms of stringency, between eligibility rules applied under TU and RD payments. The implementation of TU payments is expected to lead to higher connectivity in the landscape and the creation of conserved land parcels near occupied land parcels; however, conserving neighbouring land parcels leads to a restriction of the number of eligible land parcels and thus higher conservation costs (see the land parcel restriction effect in Drechsler *et al.*, 2010). For a given conservation budget, this can result in the conservation of a smaller number of land parcels and thus potentially lower conservation results compared to RD payments. Indeed, it can be observed that the degree of clustering of conserved land parcels is higher under TU payments when compared to RD payments for low budget levels (see Figures III.11 to III.13). This difference in terms of the degree of clustering of conserved land parcels between TU and RD payments is alleviated with increasing conservation budgets (see Table III.7). Besides, Figures III.5 to III.7 show that the number of land parcels conserved under TU payments is always lower than the number of conserved land parcels under RD payments for low levels of conservation budget (for $Clim_stab = 5$ and $\sigma = 0.1$).

Finally, results of both binary logit and multiple regression models point to the significant cost-effectiveness gains allowed by TRD payments compared to TU and RD payments (Tables III.3(d), III.3(e), III.4(d) and III.4(e)). TRD payments achieve strictly better conservation results under low and intermediate levels of conservation budget compared to TU and RD payments. We could observe that the three payments schemes lead to similar conservation results for higher budget levels. Among the four considered payment schemes, the TRD payment scheme is the one showing the best performance in terms of cost-effectiveness, and this, despite the existence of significant monitoring/cost-effectiveness trade-offs.

The better cost-effectiveness performance of more sophisticated payment schemes for low conservation budgets is due to the eligibility rules related to these payments. These eligibility rules allow — in the broad sense — to (more) effectively target conservation payments and ensure the conservation of at least some climatically suitable and occupied land parcels. For higher conservation budgets this advantage decreases since enough

money is available to conserve a sufficiently high number of climatically suitable land parcels and reach a sufficient degree of connectivity between conserved land parcels under all payment schemes.

Level of heterogeneity of conservation costs in the landscape σ

Results of the regression model show that an increase of cost-heterogeneity σ in the landscape decreases the relative cost-effectiveness of TU payments compared to U payments. Table III.4(b) shows that the difference between the mean extinction time in the landscape obtained under TU payments and the one obtained under U payments decreases with an increase in σ , all else being equal (Table III.4(b)).

Similar results are obtained for the RD-U comparison: Table III.4(a) indicates that the relative cost-effectiveness of RD payments compared to U payments decreases with an increase in the level of cost heterogeneity, but to a lesser extent. This result can also be explained by the difference, in terms of stringency, between eligibility rules applied under RD and U payments. Nevertheless, it should be mentioned that these results are not captured by the logit models considering the RD-U and TU-U pairwise comparisons (Tables III.3(a) and III.3(b)).

The difference in terms of the number of conserved land parcels between TU and RD payments significantly decreases with an increase in σ (Table III.5). This result is due to the eligibility rule applied under TU payments and its stringency compared to the stringency of the eligibility rule applied under RD payments. As it has been shown by Drechsler *et al.*, (2010), “the land parcel restriction effect is largest when the cost-heterogeneity among land parcels in the landscape is high” and an increase in the *Sigma* parameter thus leads to an increase in the cost of connectivity. This phenomenon thus leads to the conservation of fewer land parcels under TU payments compared to RD payments, but it does not have any impact on the relative cost-effectiveness of TU and RD payments (Tables III.3(c) and III.4(c)).

Considering the TRD-TU comparison, results of the binary logit model suggest that an increase in σ leads to a significant decrease in the probability to obtain a strictly better conservation result under TRD payments compared to TU payments (Table III.3(d)). Thus, the relative cost-effectiveness of TRD payments is reduced by an increase of σ

compared to TU payments. Again, this result could be related to the higher stringency of the eligibility rule applied under TRD payments compared to TU payments. However, this impact of σ on the relative cost-effectiveness of TRD payments is not captured by the regression model.

Comparing TRD and RD payments, Table III.4(e) indicates that an increase in σ leads to a significant decrease in the difference in terms of mean extinction time between TRD and RD payments. Besides, Table III.11 shows that the difference in terms of the number of conserved land parcels between TRD and RD payments significantly decreases with an increase in σ . However, these results unexpectedly do not concur with results obtained with the logistic regression model which shows that an increase in conservation costs heterogeneity leads to a significant increase in the probability to obtain strictly better conservation results under TRD payments compared to RD payments (Table III.3(e)).

6.2 Impact of variations in payment schemes characteristics

Level of monitoring and transaction costs m and T

We could observe that the implementation of targeted and/or differentiated payment schemes involves significant monitoring cost/cost-effectiveness trade-offs. Results obtained for both logit and regression models show that coefficients associated to the impact of monitoring and transaction cost variables are negative and significantly different from zero (i.e., *Transaction_cost_10_UD*, *Transaction_cost_20_UD*, *Monitoring_cost_0.01_TU*, etc.). Values of coefficients associated to both variables are large and show the same magnitude when considering the impact of monitoring and transaction cost (Tables III.3(a) to III.3(e) and Tables III.4(a) to III.4(e)). These results hold for the five pairwise comparisons. Hence, an increase in monitoring and/or transaction costs significantly and negatively impacts the relative cost-effectiveness performance of the reference payment scheme compared to the second considered payment scheme and vice-versa.

Table III.3: Estimation results – Mean extinction time – Binary logit model

(a) Regionally differentiated payments (RD) vs. uniform payments (U)

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	3.336	***	0.171	19.499	< 2e-16	0.749
<i>Disp_dist</i>	-0.658	***	0.028	-23.238	< 2e-16	-0.148
<i>Clim_stab</i>	-0.084	***	0.021	-3.996	6.45e-05	-0.019
<i>Budget</i>	-0.019	***	0.002	-10.568	< 2e-16	-0.004
<i>Transaction_cost_10_RD</i>	-1.225	***	0.092	-13.338	< 2e-16	-0.251
<i>Transaction_cost_20_RD</i>	-2.330	***	0.101	-23.159	< 2e-16	-0.444
<i>Clim_threshold_0.5_RD</i>	0.417	***	0.091	4.608	4.07e-06	0.096
<i>Clim_threshold_0.7_RD</i>	0.603	***	0.096	6.269	3.64e-10	0.140
McFadden pseudo R^2	0.249					

(b) Targeted uniform payments (TU) vs. uniform payments (U)

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	3.449	***	0.305	11.316	< 2e-16	0.608
<i>Disp_dist</i>	-1.092	***	0.065	-16.893	< 2e-16	-0.193
<i>Clim_stab</i>	0.075	*	0.040	1.863	0.062	0.013
<i>Budget</i>	-0.010	***	0.003	-3.202	0.001	-0.002
<i>Monitoring_cost_0.05_TU</i>	-1.645	***	0.183	-8.964	< 2e-16	-0.248
<i>Monitoring_cost_0.1_TU</i>	-2.348	***	0.193	-12.182	< 2e-16	-0.349
McFadden pseudo R^2	0.337					

(c) Regionally differentiated payments (RD) vs. targeted uniform payments (TU)

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	0.885	***	0.121	7.283	3.26e-13	0.042
<i>Disp_dist</i>	0.343	***	0.019	18.149	< 2e-16	0.016
<i>Clim_stab</i>	-0.262	***	0.015	-17.036	< 2e-16	-0.013
<i>Budget</i>	-0.095	***	0.002	-41.236	< 2e-16	-0.005
<i>Transaction_cost_10_RD</i>	-1.324	***	0.065	-20.356	< 2e-16	-0.055
<i>Transaction_cost_20_RD</i>	-3.208	***	0.091	-35.372	< 2e-16	-0.136
<i>Clim_threshold_0.5_RD</i>	0.201	***	0.070	2.864	0.004	0.010
<i>Clim_threshold_0.7_RD</i>	0.459	***	0.071	6.493	8.40e-11	0.024
<i>Monitoring_cost_0.05_TU</i>	1.355	***	0.079	17.213	< 2e-16	0.083
<i>Monitoring_cost_0.1_TU</i>	2.677	***	0.083	32.398	< 2e-16	0.219
McFadden pseudo R^2	0.439					

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.3: Estimation results – Mean extinction time – Binary logit model

(d) Targeted regionally differentiated payments (TRD) vs. targeted uniform payments (TU)

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	2.433	***	0.127	19.183	< 2e-16	0.257
<i>Disp_dist</i>	-0.164	***	0.017	-9.629	< 2e-16	-0.017
<i>Clim_stab</i>	-0.323	***	0.015	-21.785	< 2e-16	-0.034
<i>Budget</i>	-0.102	***	0.002	-48.561	< 2e-16	-0.011
<i>Sigma</i>	-0.925	*	0.477	-1.941	0.052	-0.098
<i>Transaction_cost_10_TRD</i>	-1.856	***	0.056	-32.929	< 2e-16	-0.213
<i>Monitoring_cost_0.05_TRD</i>	-0.434	***	0.051	-8.425	< 2e-16	-0.046
<i>Clim_threshold_0.5_TRD</i>	0.623	***	0.063	9.915	< 2e-16	0.071
<i>Clim_threshold_0.7_TRD</i>	0.934	***	0.064	14.514	< 2e-16	0.116
<i>Monitoring_cost_0.05_TU</i>	2.181	***	0.073	29.898	< 2e-16	0.304
<i>Monitoring_cost_0.1_TU</i>	3.819	***	0.082	46.554	< 2e-16	0.604
McFadden pseudo R^2	0.468					

(e) Targeted regionally differentiated payments (TRD) vs. regionally differentiated payments (RD)

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	3.112	***	0.063	49.013	< 2e-16	0.545
<i>Disp_dist</i>	-1.041	***	0.012	-89.951	< 2e-16	-0.182
<i>Budget</i>	-0.042	***	0.001	-52.853	< 2e-16	-0.007
<i>Sigma</i>	0.440	*	0.228	1.929	0.054	0.077
<i>Transaction_cost_10_TRD</i>	-1.061	***	0.027	-39.495	< 2e-16	-0.188
<i>Monitoring_cost_0.05_TRD</i>	-0.170	***	0.026	-6.531	6.53e-11	-0.030
<i>Clim_threshold_0.5_TRD</i>	0.170	***	0.031	5.440	5.33e-08	0.030
<i>Clim_threshold_0.7_TRD</i>	0.254	***	0.033	7.756	8.73e-15	0.046
<i>Transaction_cost_10_RD</i>	0.934	***	0.034	27.471	< 2e-16	0.176
<i>Transaction_cost_20_RD</i>	2.206	***	0.036	61.765	< 2e-16	0.433
<i>Clim_threshold_0.5_RD</i>	-0.509	***	0.031	-16.273	< 2e-16	-0.085
<i>Clim_threshold_0.7_RD</i>	-1.011	***	0.033	-30.483	< 2e-16	-0.158
McFadden pseudo R^2	0.349					

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.4: Estimation results – Mean extinction time – Multiple linear regression

(a) Regionally differentiated payments (RD) vs. uniform payments (U)					
Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	12.969	***	1.469	8.827	< 2e-16
<i>Disp_dist</i>	-3.886	***	0.217	-17.927	< 2e-16
<i>Clim_stab</i>	-0.539	***	0.195	-2.766	0.006
<i>Budget</i>	0.348	***	0.016	21.577	< 2e-16
<i>Sigma</i>	-12.149	**	5.847	-2.078	0.038
<i>Transaction_cost_10_RD</i>	-18.343	***	0.791	-23.185	< 2e-16
<i>Transaction_cost_20_RD</i>	-31.066	***	0.954	-32.564	< 2e-16
<i>Clim_threshold_0.5_RD</i>	5.473	***	0.757	7.226	5.858e-13
<i>Clim_threshold_0.7_RD</i>	9.856	***	0.910	10.834	< 2e-16
Multiple R^2	0.319			Adjusted R^2	0.318
(b) Targeted uniform payments (TU) vs. uniform payments (U)					
Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	27.619	***	2.484	11.120	< 2e-16
<i>Disp_dist</i>	-9.264	***	0.432	-21.467	< 2e-16
<i>Budget</i>	0.422	***	0.028	15.243	< 2e-16
<i>Sigma</i>	-18.789	*	10.986	-1.710	0.087
<i>Monitoring_cost_0.05_TU</i>	-14.638	***	1.480	-9.889	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	-27.475	***	1.704	-16.127	< 2e-16
Multiple R^2	0.392			Adjusted R^2	0.39
(c) Regionally differentiated payments (RD) vs. targeted uniform payments (TU)					
Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	-17.002	***	1.136	-14.969	< 2e-16
<i>Disp_dist</i>	6.073	***	0.162	37.431	< 2e-16
<i>Clim_stab</i>	-0.596	***	0.154	-3.875	1.071e-04
<i>Budget</i>	-0.103	***	0.013	-7.820	5.661e-15
<i>Transaction_cost_10_RD</i>	-14.188	***	0.638	-22.224	< 2e-16
<i>Transaction_cost_20_RD</i>	-29.077	***	0.667	-43.566	< 2e-16
<i>Clim_threshold_0.5_RD</i>	4.230	***	0.606	6.975	3.193e-12
<i>Clim_threshold_0.7_RD</i>	8.783	***	0.646	13.603	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	13.904	***	0.611	22.766	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	29.121	***	0.655	44.440	< 2e-16
Multiple R^2	0.292			Adjusted R^2	0.292

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.4: Estimation results – Mean extinction time – Multiple linear regression

(d) Targeted regionally differentiated payments (TRD) vs. targeted uniform payments (TU)

Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	9.295	***	0.726	12.797	< 2e-16
<i>Clim_stab</i>	-0.856	***	0.114	-7.536	5.105e-14
<i>Budget</i>	-0.222	***	0.011	-20.286	< 2e-16
<i>Transaction_cost_10_TRD</i>	-14.130	***	0.403	-35.039	< 2e-16
<i>Monitoring_cost_0.05_TRD</i>	-6.779	***	0.395	-17.149	< 2e-16
<i>Clim_threshold_0.5_TRD</i>	6.239	***	0.451	13.830	< 2e-16
<i>Clim_threshold_0.7_TRD</i>	9.749	***	0.523	18.624	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	14.749	***	0.392	37.593	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	32.688	***	0.557	58.686	< 2e-16
Multiple R^2	0.324			Adjusted R^2	0.324

(e) Targeted regionally differentiated payments (TRD) vs. regionally differentiated payments (RD)

Variable	Coef.		Std. error	t value	p value
Intercept	21.489	***	0.740	29.054	< 2e-16
Disp_dist	-5.882	***	0.094	-62.246	< 2e-16
Clim_stab	0.701	***	0.090	7.820	5.42e-15
Budget	-0.144	***	0.008	-17.706	< 2e-16
Sigma	-4.891	**	2.280	-2.145	0.032
Transaction_cost_10_TRD	-15.116	***	0.288	-52.552	< 2e-16
Monitoring_cost_0.05_TRD	-7.106	***	0.284	-25.023	< 2e-16
Clim_threshold_0.5_TRD	6.652	***	0.331	20.107	< 2e-16
Clim_threshold_0.7_TRD	10.727	***	0.366	29.347	< 2e-16
Transaction_cost_10_RD	15.496	***	0.325	47.736	< 2e-16
Transaction_cost_20_RD	35.155	***	0.386	91.028	< 2e-16
Clim_threshold_0.5_RD	-3.981	***	0.339	-11.755	< 2e-16
Clim_threshold_0.7_RD	-8.328	***	0.351	-23.706	< 2e-16
Multiple R^2	0.298			Adjusted R^2	0.298

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Payment eligibility threshold for climate suitability γ_e

Comparing TU-RD and TRD-RD payments, we find that an increase in the payment eligibility threshold for climate suitability γ_e related to the RD payment decreases

the cost-effectiveness of TU payments compared to RD payments. Indeed, Tables III.3(c) and III.3(e) show that coefficients associated to the *Clim.threshold_0.5_RD* and *Clim.threshold_0.7_RD* variables are negative and significantly different from zero. This means that an increase in γ_e leads to a significant decrease in the probability to achieve strictly better conservation results under TU compared to RD payments. Marginal effects associated to these variables are large. Similar results are obtained for the regression models (see Tables III.4(c) and III.4(e)).

Two reasons explain this result. First, an increase in γ_e means that only more climatically suitable land parcels are conserved. Second, the size of the eligible area for conservation payments shrinks with an increase in γ_e : conservation is thus restricted to a smaller climatically suitable region of the landscape, which indirectly increases connectivity between conserved land parcels.

6.3 Impact in variations in ecological variables

Species mean dispersal distance d (Disp_dist)

Results of both logit and regression models highlight the higher cost-effectiveness of targeted payments (TU and TRD payments) compared to RD and U payments for the conservation of species with low mean dispersal distance. The coefficient related to the dispersal distance variable (i.e., Disp_dist) shows a negative sign and is significantly different from zero (Tables III.3(b) and III.3(e)). Moreover, the marginal effect associated to the latter variable is large. This means that the probability to achieve strictly better conservation results under TU (resp. TRD) compared to U (resp. RD) payments significantly decreases with an increase in the target species' mean dispersal distance. When considering the RD-TU comparison, results point the higher cost-effectiveness of RD payments compared to TU payments for the conservation of species showing higher dispersal ability (Table III.3(c)). The estimation of a logit model based on the TU-RD comparison also points to the higher cost-effectiveness of TU payments compared to RD payments if species with a low mean dispersal distance are conserved (Appendix 9.4, Table III.10).

RD payments enable cost-effectiveness gains compared to U payments for the conservation of species showing a low mean dispersal distance (Table III.3(a) and III.4(a)).

This result can be explained by the fact that, under RD payments, only land parcels satisfying the climate suitability rule are eligible for conservation. Conservation is thus restricted to a smaller climatically suitable region of the landscape, which — as explained earlier — indirectly increases connectivity in the landscape and favours species showing a poor dispersal ability.

Results obtained from the multiple linear regression models also point to the higher cost-effectiveness of TU payments for the conservation of short dispersers (Tables III.4(b), III.4(c) and III.4(d)). Considering the TRD-TU comparison, results of the logit model underline the higher cost-effectiveness of TRD payments for the conservation of species showing low mean dispersal distance compared to TU payments. However, this result is not captured by the regression model. As above, the higher cost-effectiveness of targeted payments (i.e., TU and TRD) for the conservation of short dispersers is due to their incentive to generate spatially cluster conserved land parcels. This is expected to result in higher connectivity in the landscape, which is favourable towards species with low dispersal abilities. Such results seem to point out the existence of connectivity/area trade-offs under climate change.

The existence of such connectivity-area trade-offs under climate change is confirmed by the following results. Comparing TU and RD payments in terms of the number of climatically suitable land parcels conserved (Figures III.8 to III.10 and Table III.6), we observe that the difference between the number of climatically suitable land parcels conserved under TU payments and RD payments significantly decreases with an increase in the dispersal distance of the species. Moreover, in Figures III.8 to III.10 the number of climatically suitable land parcels conserved under RD payments is always higher than the number of climatically suitable land parcels conserved under TU payments for intermediate to high levels of the payment eligibility threshold for climate suitability (i.e., $\gamma_e = 0.5$ and $\gamma_e = 0.7$). Thus, implementing RD payments allows to conserve a larger climatically suitable area compared to TU payments. Besides, results show that the higher the species' dispersal ability, the larger the difference between the number of climatically suitable land parcels conserved between RD and TU payments.

Besides, Figures III.11 to III.16 show that TU payments generally lead to a higher degree of clustering of conserved and climatically suitable land parcels compared to RD

payments for low and intermediate values of γ_e ($\gamma_e = 0.3$ and $\gamma_e = 0.5$). Moreover, the degree of clustering of conserved land parcels is always higher under TU payments compared to RD payments for short dispersers (i.e., $Disp_dist = 1$) compared to RD payments, except for $\gamma_e = 0.7$ and high level of conservation budget. Finally, the difference in terms of the degree of clustering of conserved and climatically suitable land parcels between TU and RD payments significantly decreases with an increase in species' mean dispersal distance (Table III.7 and III.8).

Table III.5: Targeted uniform payments (TU) vs. regionally differentiated payments (RD) – Number of conserved land parcels – Multiple linear regression

Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	-1.214	***	0.106	-11.486	< 2e-16
<i>Disp_dist</i>	0.159	***	0.014	11.428	< 2e-16
<i>Budget</i>	-0.003	*	0.001	-1.819	0.069
<i>Clim_stab</i>	0.034	***	0.013	2.683	0.007
<i>Sigma</i>	-2.342	***	0.415	-5.647	1.676e-08
<i>Transaction_cost_10_RD</i>	7.588	***	0.050	153.044	< 2e-16
<i>Transaction_cost_20_RD</i>	15.791	***	0.059	267.298	< 2e-16
<i>Clim_threshold_0.5_RD</i>	0.965	***	0.051	18.904	< 2e-16
<i>Clim_threshold_0.7_RD</i>	1.427	***	0.059	24.393	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	-7.527	***	0.046	-163.372	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	-15.619	***	0.060	-261.776	< 2e-16
Multiple R^2	0.938			Adjusted R^2	0.938

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.6: Targeted uniform payments (TU) vs. regionally differentiated payments (RD) – Number of climatically suitable conserved land parcels – Multiple linear regression

Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	3.217	***	0.170	18.960	< 2e-16
<i>Disp_dist</i>	-0.096	***	0.033	-2.913	0.004
<i>Clim_stab</i>	0.175	***	0.022	8.072	8.901e-16
<i>Transaction_cost_10_RD</i>	6.395	***	0.112	56.952	< 2e-16
<i>Transaction_cost_20_RD</i>	13.400	***	0.114	117.153	< 2e-16
<i>Clim_threshold_0.5_RD</i>	-7.105	***	0.113	-63.024	< 2e-16
<i>Clim_threshold_0.7_RD</i>	-6.055	***	0.110	-54.969	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	-6.349	***	0.102	-62.482	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	-13.188	***	0.117	-112.269	< 2e-16
Multiple R^2	0.881			Adjusted R^2	0.881

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.7: Targeted uniform payments (TU) vs. regionally differentiated payments (RD) – Degree of clustering of conserved land parcels – Multiple linear regression

Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	1.541	***	0.035	43.633	< 2e-16
<i>Disp_dist</i>	-0.126	***	0.006	-21.801	< 2e-16
<i>Clim_stab</i>	0.009	**	0.004	2.396	0.017
<i>Budget</i>	-0.008	***	0.001	-16.383	< 2e-16
<i>Transaction_cost_10_RD</i>	0.623	***	0.018	34.793	< 2e-16
<i>Transaction_cost_20_RD</i>	1.305	***	0.020	64.739	< 2e-16
<i>Clim_threshold_0.5_RD</i>	-0.588	***	0.018	-32.572	< 2e-16
<i>Clim_threshold_0.7_RD</i>	-0.918	***	0.022	-42.383	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	-0.643	***	0.020	-32.878	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	-1.355	***	0.018	-74.297	< 2e-16
Multiple R^2	0.739			Adjusted R^2	0.739

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Table III.8: Targeted uniform payments (TU) vs. regionally differentiated payments (RD) – Degree of clustering of climatically suitable conserved land parcels – Multiple linear regression

Variable	Coef.		Std. error	t value	p value
<i>Intercept</i>	1.451	***	0.034	42.324	< 2e-16
<i>Disp_dist</i>	-0.128	***	0.006	-22.857	< 2e-16
<i>Clim_stab</i>	0.019	***	0.004	5.092	3.7e-07
<i>Budget</i>	-0.007	***	0.0005	-15.429	< 2e-16
<i>Transaction_cost_10_RD</i>	0.612	***	0.017	35.311	< 2e-16
<i>Transaction_cost_20_RD</i>	1.284	***	0.020	65.652	< 2e-16
<i>Clim_threshold_0.5_RD</i>	-0.676	***	0.017	-38.678	< 2e-16
<i>Clim_threshold_0.7_RD</i>	-1.002	***	0.021	-47.283	< 2e-16
<i>Monitoring_cost_0.05_TU</i>	-0.624	***	0.019	-32.846	< 2e-16
<i>Monitoring_cost_0.1_TU</i>	-1.313	***	0.018	-73.537	< 2e-16
Multiple R^2	0.750			Adjusted R^2	0.750

***, ** and * stand for 1%, 5% and 10% significance levels respectively

Level of climate stability in the landscape $Clim_stab$

Finally, we find that RD payments are more cost-effective than U and TU payments for conservation under rapid climate change. Indeed, in Tables III.3(a) and III.3(c) the coefficient related to the *Clim_stab* variable shows a negative sign and is significantly different from zero. This means that an increase in climate stability leads to a decrease in the probability to achieve strictly better conservation results under RD payments compared to U and TU payments. Results obtained from the linear multiple regression models show similar results (Tables III.4(a) and III.4(c)). Considering the TRD-RD comparison, we observe that the difference between the mean extinction time in the landscape obtained under TRD payments and the one obtained under RD payments significantly increases with an increase in climate stability in the landscape (Table III.4(e)). Besides, it can be observed from Table III.3(d) and III.4(d) that TRD payments perform better than TU payments under rapid climate change.

This result, as mentioned earlier, can be due to the conservation of a higher number of climatically suitable land parcels under differentiated RD compared to TU payments. This difference is deepened under rapid climate change as it can be observed from Table III.6. Besides, advantages in terms of the degree of clustering of conserved land parcels brought by TU payments are reduced by a decrease in the level of climate stability as

it can be observed from Table III.7 and III.8.

7 Discussion and Conclusion

We developed an ecological-economic model to investigate the cost-effectiveness of several payment design options showing varying degrees of payments' differentiation and targeting in a landscape whose dynamics is driven by climate change. We studied the impact of changes in economic, ecological and climate parameters on the relative cost-effectiveness performance of four design options: uniform payments, regionally differentiated payments (RD), targeted uniform payments (TU) and targeted regionally differentiated payments (TRD). This study is the first work investigating the cost-effectiveness of the design of payments for biodiversity conservation under climate change on a conceptual level.

Our results suggest that significant cost-effectiveness gains are feasible by the implementation of targeted and differentiated payments under climate change compared to simple uniform untargeted payments. We also show that classic connectivity/area trade-offs which were highlighted by Drechsler (2011) and Johst *et al.*, (2011) in dynamic landscapes are also observed under climate change. Besides, according to our results the implementation of targeted payments appears to be better suited to the conservation of species showing poor dispersal ability. By contrast, the relative cost-effectiveness of regionally differentiated payments increases compared to targeted payments for the conservation of species showing better dispersal ability. Finally, we show that the advantages brought by targeted payments in terms of connectivity of the conservation network are decreased by a decrease in the level of climate stability in the landscape. In this case, the relative cost-effectiveness of targeted payments appears to be undermined relatively to differentiated payments. This last result is consistent with those obtained by Drechsler (2017) who shows that the performance of an output-based payment granted based on species presence increases with increasing viability and decreasing dispersal ability of the target species.

This model is based on several assumptions, which are worth discussing. First, we assume that opportunity costs of conservation are not spatially correlated and insensitive to climate change. Nevertheless, it has been shown that climate change will induce

a temporal variation in rents of existing land-use and management practices (Olesen and Bindi, 2002). This should thus influence landowner's land-use choices and thus have an influence on opportunity costs of conservation. Considering spatial correlation of opportunity cost related to climate suitability would affect our results in two different ways depending on the sign of the correlation between opportunity cost and climate suitability. Considering that climate-suitability and opportunity costs are positively correlated would not change our results. In this case, the level of conservation costs would be higher within the climatically suitable zone which would lead to an increase of conservation costs for targeted and differentiated payments. However, this would not undermine the better cost-effectiveness of targeted and differentiated payment schemes compared to uniform payments since, for uniform payments, conservation would be carried out outside the climatically suitable zone first. By contrast, considering a negative correlation between climate suitability and opportunity cost and conservation would lead conservation to be carried out within the climatically suitable zone first for all payment schemes. In this case, cost-effectiveness gains of more sophisticated schemes would be undermined.

Second, we assume constant transaction and total monitoring costs over time. It could be argued that the burden of cost should decrease in the course of climate change since the conservation planner would notice that the southern part of the landscape is no longer suitable for conservation. Accounting for this relative assumption by considering decreasing transaction and total monitoring costs would lead to even higher cost-effectiveness gains enabled by more sophisticated schemes and would not change our results.

Finally, our results are valid within the parameter space selected and it would be interesting to use our research as the basis of a more applied work involving empirical data on conservation, monitoring and transaction costs, climate scenarios as well as data on characteristics of a (several) specific target species. We leave this for future research.

This work is based on a conceptual approach and the direct relevance of our results in terms of designing specific conservation policy in practice is admittedly limited. However, our research provides the first generalised insight into the design of cost-effective incentive payments for conservation under climate change. Besides, the significant impact of conservation context variables on the cost-effectiveness of considered schemes

seems to point the need to carefully consider this conservation context and characteristics of the target species when designing conservation payments.

8 Acknowledgments

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9 Appendix

9.1 Definition of variables and descriptive statistics

Table III.9: Definition of variables and descriptive statistics (1/2)

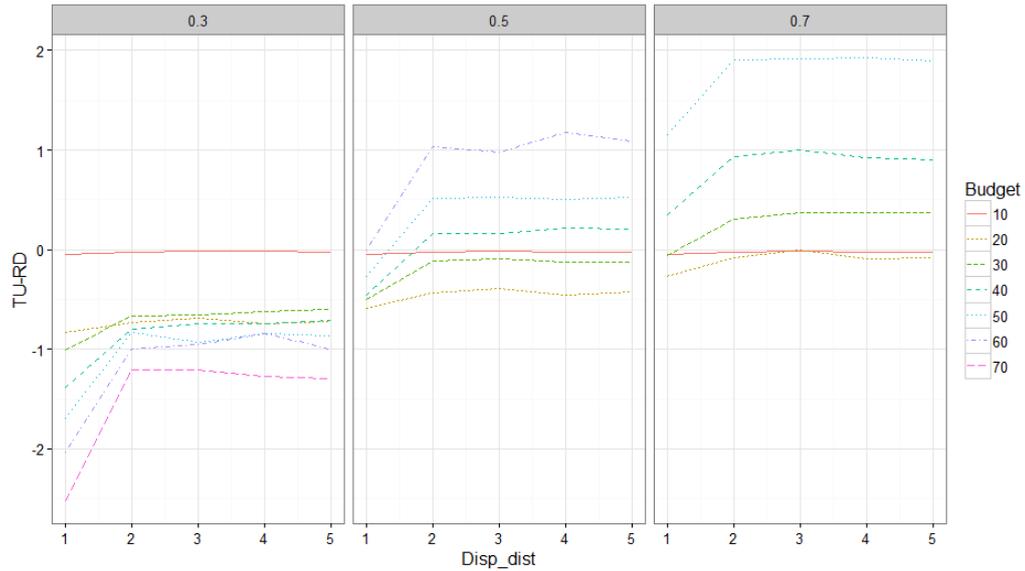
Variable	Description	Min	Max	Mean	N
U	Dummy if uniform payment implemented	0	1	-	0: 13,222 1: 708
TU	Dummy if targeted uniform payment implemented	0	1	-	0: 11,973 1: 1,957
TRD	Dummy if targeted regionally differentiated payment implemented	0	1	-	0: 8,560 1: 5,370
RD	Dummy if regionally differentiated payment implemented	0	1	-	0: 8,035 1: 5,895
Targeted	Dummy if targeted payment implemented	0	1	-	0: 6,603 1: 7,327
Differentiated	Dummy if differentiated payment implemented	0	1	-	0: 2,665 1: 11,265
Sigma	Conservation cost heterogeneity in the landscape	0.05	0.3	0.120	13,930
Clim_stab	Climate stability (time frame with stable climate, in time periods)	1	7	4.15	13,930
Disp_dist	Species' mean dispersal distance	1	5	2.92	13,930
Budget	Conservation budget	10	100	40.51	13,930
Monitoring_cost	Monitoring cost (land parcel occupation)	0	0.1	0.019	13,930
Transaction_cost	Transaction cost (climate suitability)	0	20	7.131	13,930
Threshold_eligibility	Eligibility threshold value for differentiated payments in terms of climate suitability	0	0.7	0.38	13,930
Mean_extin_time	Mean time to species extinction	0	100	81.71	13,930
Prop_non_extin	Average probability of surviving the time frame of the simulation	0	1	0.765	13,930
Mean_nb_cons_patches	Average number of conserved land parcels per time period	0	91.69	30.37	13,930
Mean_nb_cons_suit_patches	Average number of climatically suitable conserved land parcels	0	59.94	25.59	2,254
Mean_connect	Average number of direct neighbours conserved per conserved land parcel	0	6.25	2.66	2,254

Table III.9: Definition of variables and descriptive statistics (2/2)

Variable	Description	Min	Max	Mean	N
Mean_suit_connect	Average number of climatically suitable direct neighbours conserved per conserved land parcel	0	5.96	2.59	2,254
TU_U_met	Difference between the species' mean extinction time obtained under TU and U payments	-99.97	86.75	1.70	1,468
RD_U_met	Difference between the species' mean extinction time obtained under RD and U payments	-99.97	80.12	-1.35	4,278
RD_TU_met	Difference between the species' mean extinction time obtained under RD and TU payments	-100	100	-2.95	14,121
TRD_TU_met	Difference between the species' mean extinction time obtained under TRD and TU payments	-85.64	100	7.15	15,065
TRD_RD_met	Difference between the species' mean extinction time obtained under TRD and RD payments	-100	100	9.14	44,173
TU_RD_nbc	Difference between the number of conserved land parcels under RD and TU payments	-21.36	19.55	-0.07	11,436
TU_RD_nbc_suit	Difference between the number of climatically suitable conserved land parcels under RD and TU payments	-22.46	22.07	-0.15	4,253
TU_RD_connect	Difference between the number of direct neighbours conserved per conserved land parcel under RD and TU payments	-2.45	3.30	0.44	4,253
TU_RD_suit_connect	Difference between the number of climatically suitable direct neighbours conserved per conserved land parcel under RD and TU payments	-2.45	3.23	0.36	4,253

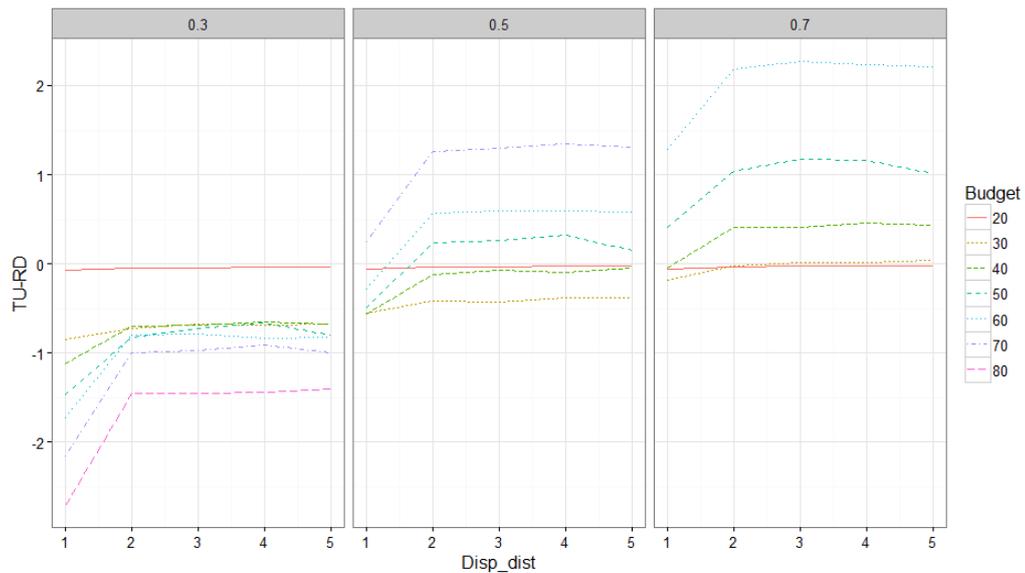
9.2 TU-RD comparison in terms of the number of land parcels conserved

Figure III.5: Comparison of the number of land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 2$, $Monitoring_cost_TU = 0.01$)



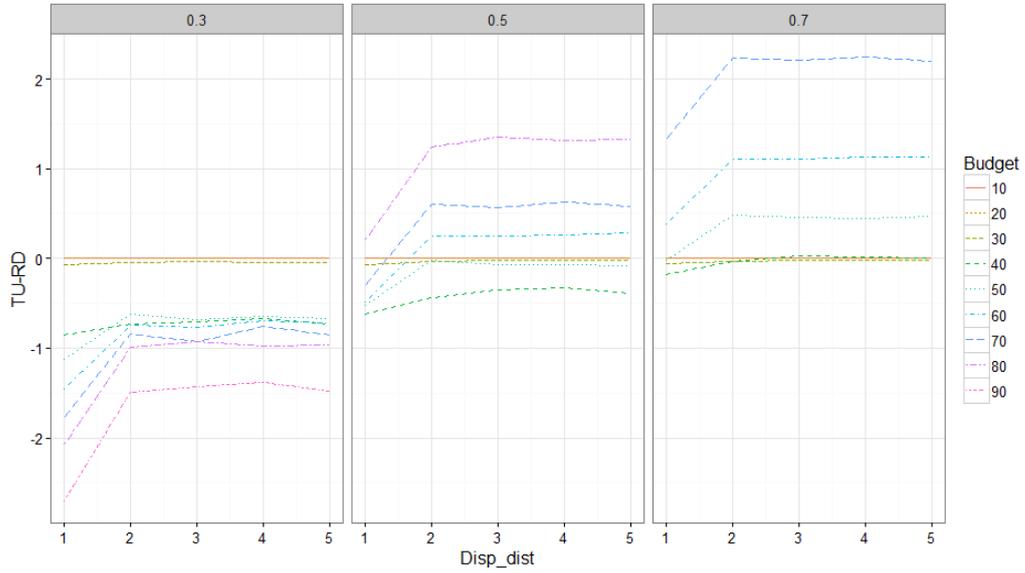
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.6: Comparison of the number of land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 10$, $Monitoring_cost_TU = 0.05$)



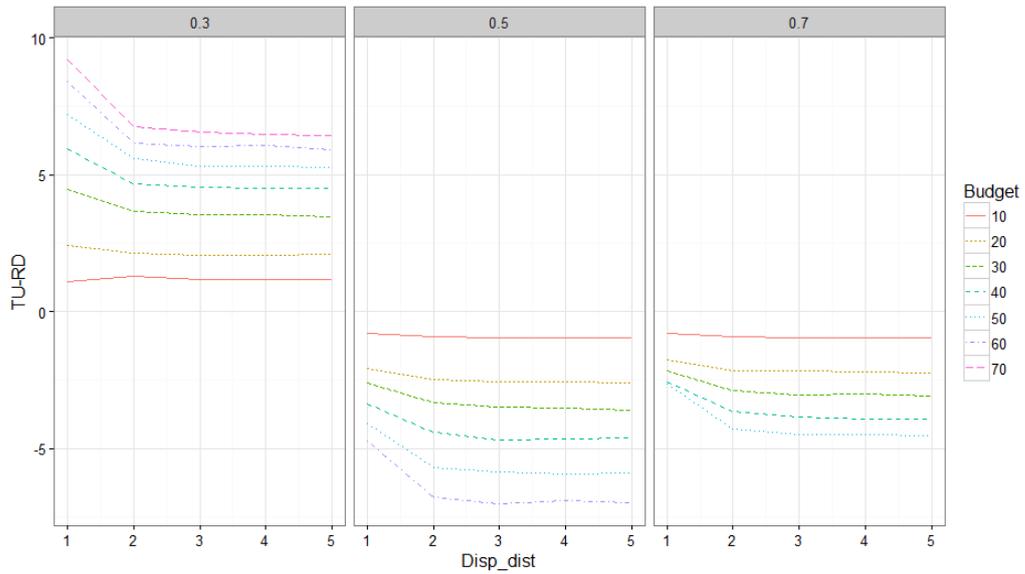
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.7: Comparison of the number of land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 20$, $Monitoring_cost_TU = 0.1$)



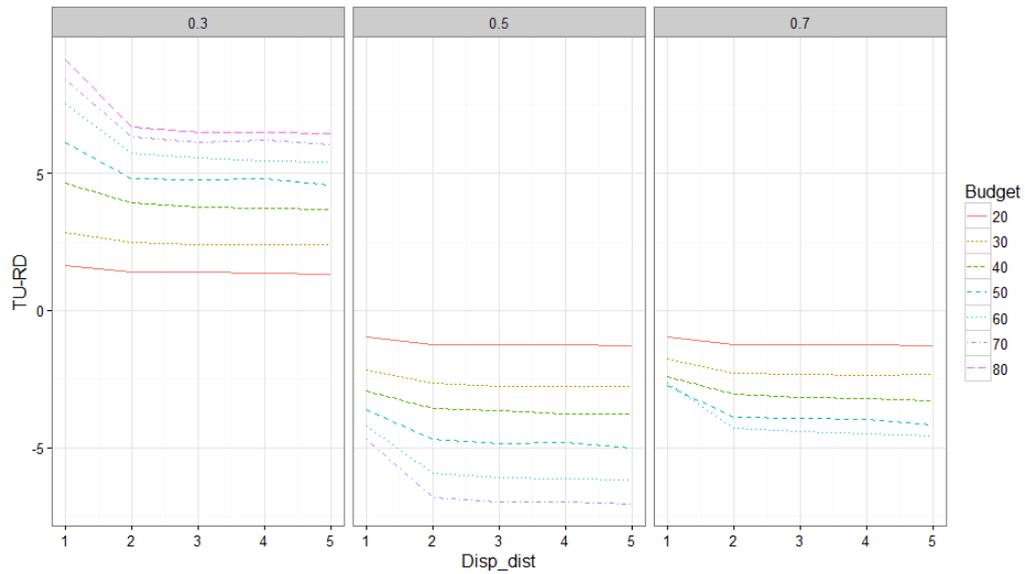
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.8: Comparison of the number of climatically suitable land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 2$, $Monitoring_cost_TU = 0.01$)



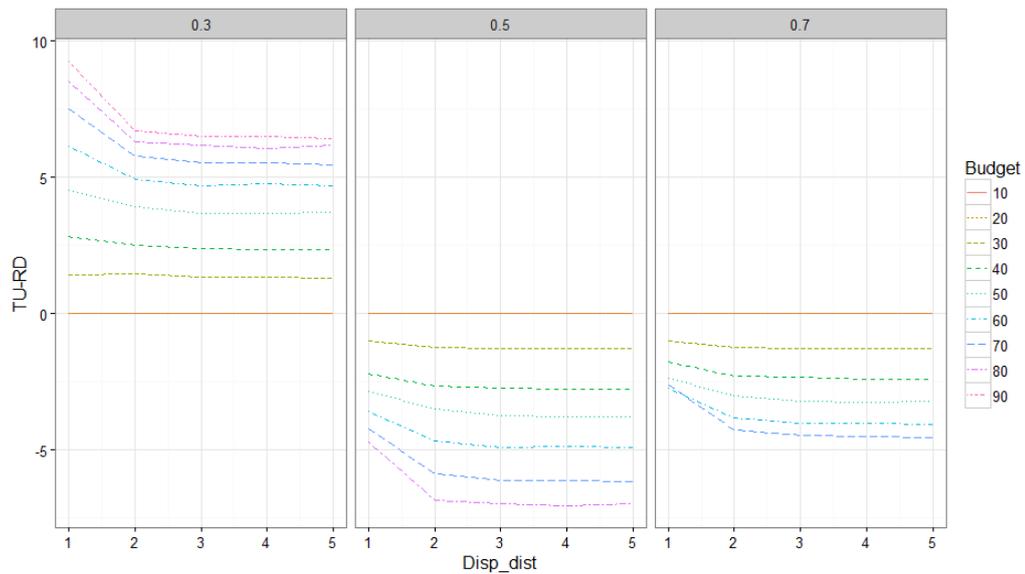
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.9: Comparison of the number of climatically suitable land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 10$, $Monitoring_cost_TU = 0.05$)



Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

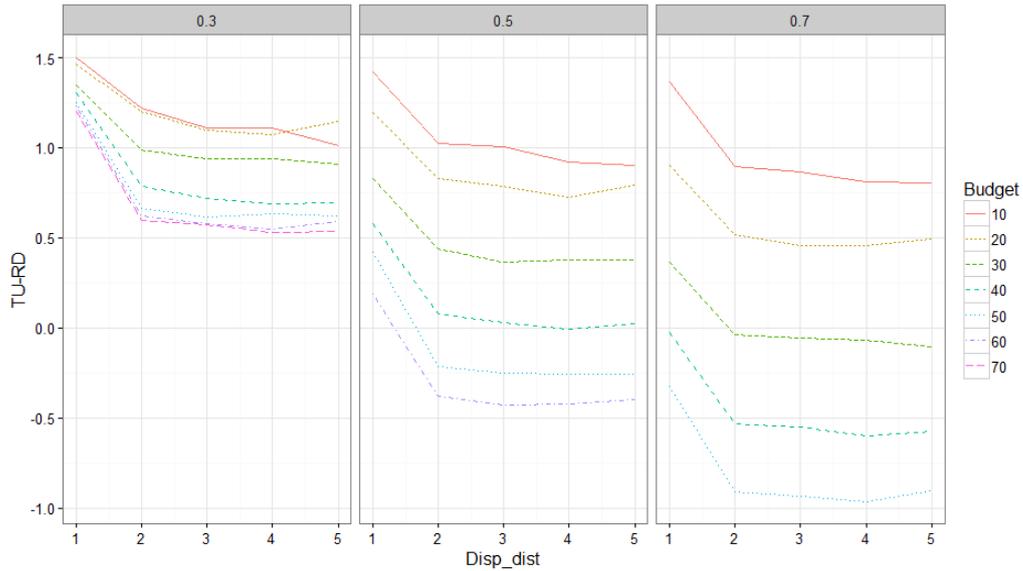
Figure III.10: Comparison of the number of climatically suitable land parcels conserved under TU and RD payments – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 20$, $Monitoring_cost_TU = 0.1$)



Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

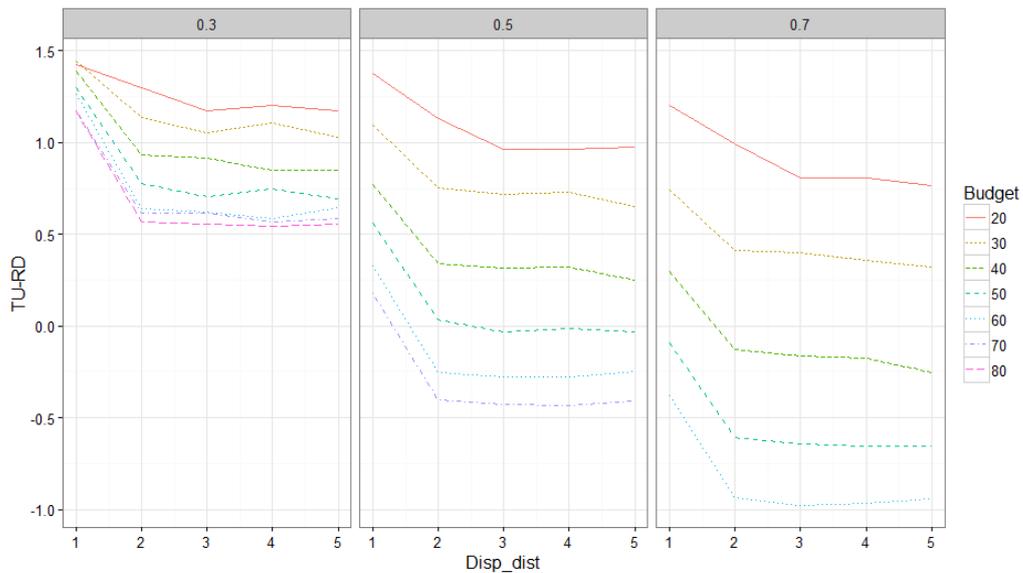
9.3 TU-RD comparison in terms of connectivity of the conservation network

Figure III.11: Comparison of TU and RD payments in terms of clustering of conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 2$, $Monitoring_cost_TU = 0.01$)



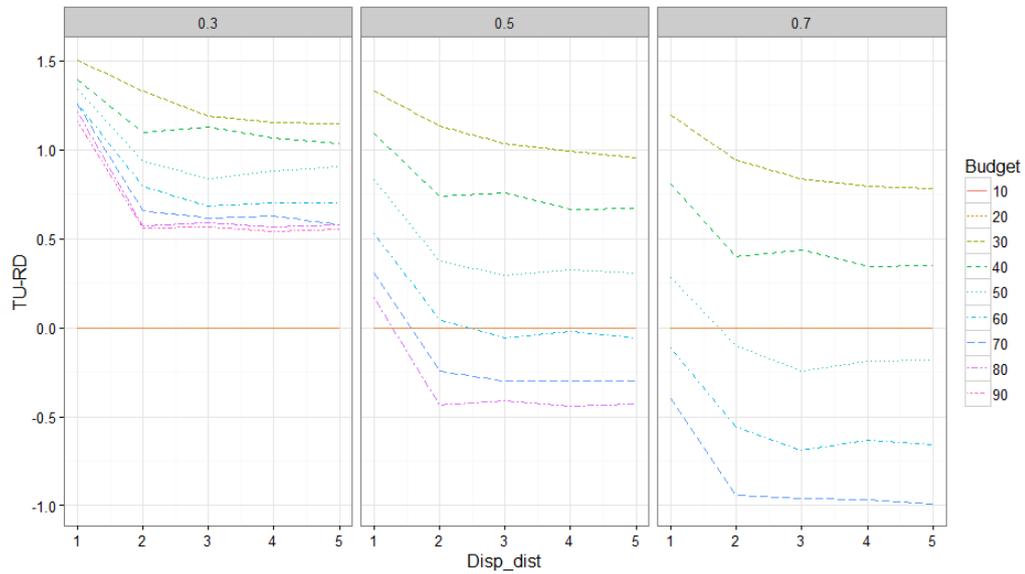
$Clim_threshold_RD = 0.3$ $Clim_threshold_RD = 0.5$ $Clim_threshold_RD = 0.7$

Figure III.12: Comparison of TU and RD payments in terms of clustering of conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 10$, $Monitoring_cost_TU = 0.05$)



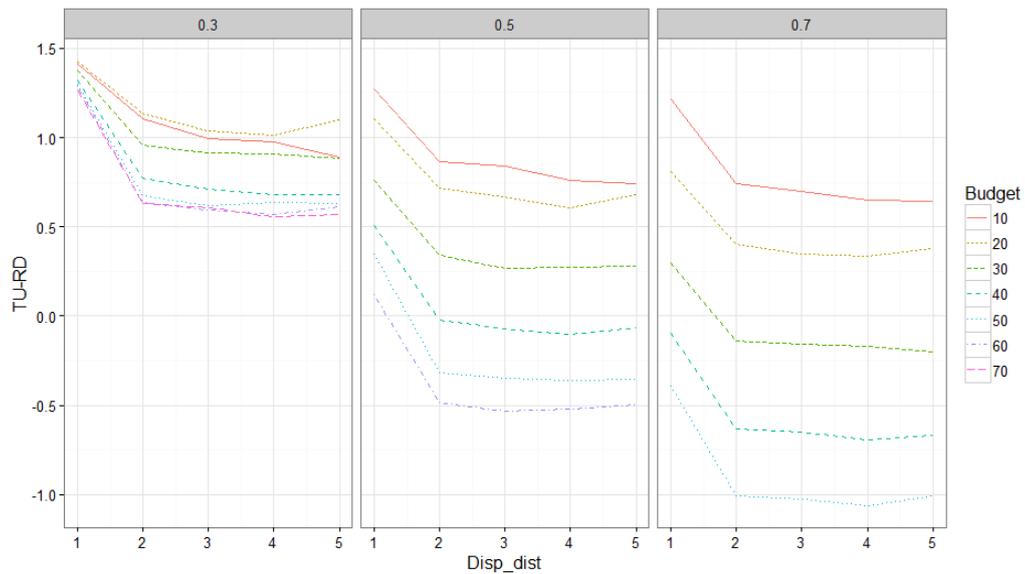
$Clim_threshold_RD = 0.3$ $Clim_threshold_RD = 0.5$ $Clim_threshold_RD = 0.7$

Figure III.13: Comparison of TU and RD payments in terms of clustering of conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 20$, $Monitoring_cost_TU = 0.1$)



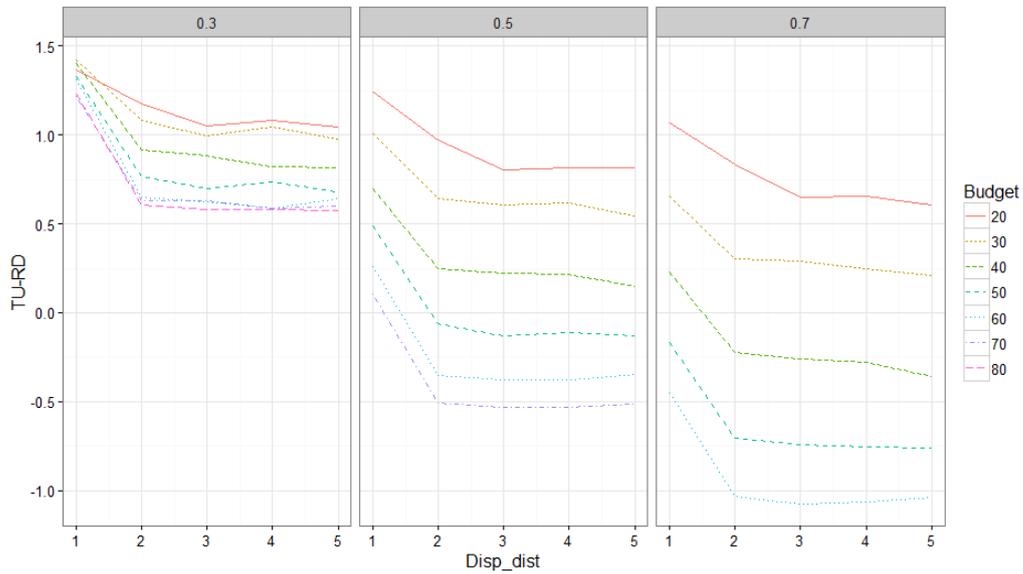
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.14: Comparison of TU and RD payments in terms of clustering of climatically suitable conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 2$, $Monitoring_cost_TU = 0.01$)



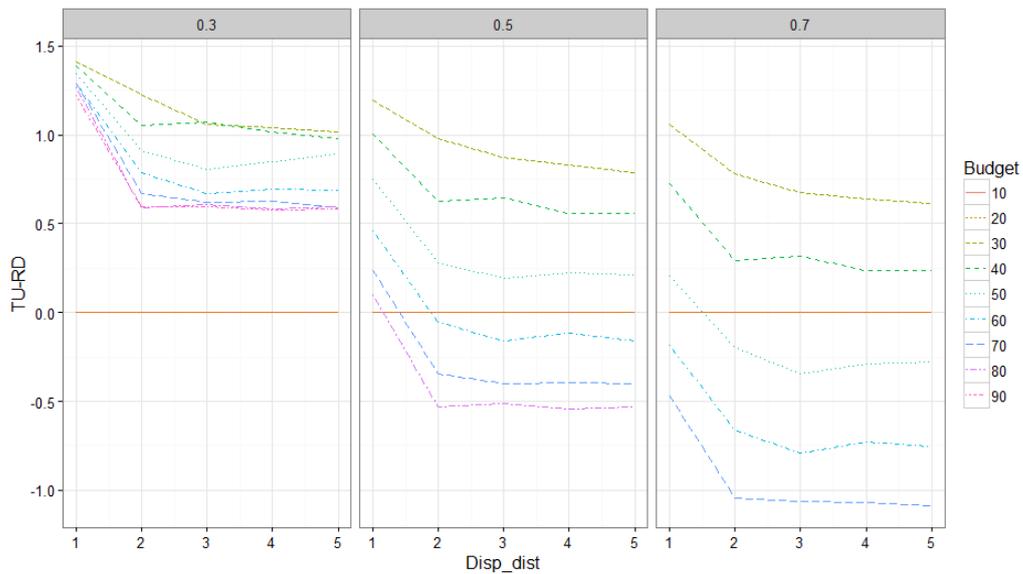
Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.15: Comparison of TU and RD payments in terms of clustering of climatically suitable conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 10$, $Monitoring_cost_TU = 0.05$)



Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

Figure III.16: Comparison of TU and RD payments in terms of clustering of climatically suitable conserved land parcels – ($Clim_stab = 5$, $Sigma = 0.1$, $Transaction_cost_RD = 20$, $Monitoring_cost_TU = 0.1$)



Clim_threshold_RD = 0.3 Clim_threshold_RD = 0.5 Clim_threshold_RD = 0.7

9.4 TU-RD comparison in terms of species' mean time to extinction

Table III.10: Targeted uniform payments (TU) vs. regionally differentiated payments (RD) Mean time to extinction – Binary Logit model

Variable	Coef.		Std. error	z value	p value	Marg. effect
<i>Intercept</i>	3.058	***	0.097	31.495	< 2e-16	0.378
<i>Disp_dist</i>	-1.246	***	0.025	-49.382	< 2e-16	-0.154
<i>Budget</i>	-0.016	***	0.001	-13.039	< 2e-16	-0.002
<i>Transaction_cost_10_RD</i>	0.586	***	0.062	9.483	< 2e-16	0.078
<i>Transaction_cost_20_RD</i>	1.422	***	0.062	22.771	< 2e-16	0.205
<i>Clim_threshold_0.5_RD</i>	-0.377	***	0.056	-6.692	2.21e-11	-0.045
<i>Clim_threshold_0.7_RD</i>	-1.018	***	0.063	-16.252	< 2e-16	-0.108
<i>Monitoring_cost_0.05_TU</i>	-1.006	***	0.058	-17.430	< 2e-16	-0.112
<i>Monitoring_cost_0.1_TU</i>	-1.878	***	0.063	-29.687	< 2e-16	-0.195
McFadden pseudo R^2	0.356					

***, ** and * stand for 1%, 5% and 10% significance levels respectively

9.5 TRD-RD comparison in terms of the number of land parcels conserved

Table III.11: Targeted regionally differentiated payments (TRD) vs. regionally differentiated payments (RD) – Number of conserved land parcels – Multiple linear regression

Variable	Coef.		Std. error	z value	p value
<i>Intercept</i>	-2.170	***	0.050	-43.553	< 2e-16
<i>Disp_dist</i>	0.107	***	0.007	16.220	< 2e-16
<i>Clim_stab</i>	-0.074	***	0.006	-13.224	< 2e-16
<i>Budget</i>	0.010	***	0.001	14.550	< 2e-16
<i>Sigma</i>	-4.902	***	0.195	-25.147	< 2e-16
<i>Transaction_cost_10_TRD</i>	-7.240	***	0.020	-363.182	< 2e-16
<i>Monitoring_cost_0.05_TRD</i>	-2.836	***	0.020	-143.491	< 2e-16
<i>Clim_threshold_0.5_TRD</i>	0.468	***	0.023	20.266	< 2e-16
<i>Clim_threshold_0.7_TRD</i>	0.551	***	0.026	21.289	< 2e-16
<i>Transaction_cost_10_RD</i>	7.666	***	0.015	502.700	< 2e-16
<i>Transaction_cost_20_RD</i>	15.665	***	0.030	527.472	< 2e-16
<i>Clim_threshold_0.5_RD</i>	0.798	***	0.024	33.891	< 2e-16
<i>Clim_threshold_0.7_RD</i>	1.287	***	0.025	50.971	< 2e-16
Multiple R^2	0.929			Adjusted R^2	0.929

***, ** and * stand for 1%, 5% and 10% significance levels respectively

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Conclusion

The aim of this thesis was to investigate the definition of efficient incentive payments for biodiversity conservation from both an empirical and a theoretical perspective.

Main results:

In the first chapter, we empirically assessed the cost-effectiveness of N2K contracts implemented in French forests through an *ex ante* analysis. The results we obtain are of direct policy relevance and allow us to provide several recommendations for the implementation of future contracts.

First, our results underline the quality of N2K site-specific management plans which ensure the coherent implementation of the various types of conservation measures. Besides, these site-specific management plans provide a useful framework for the protection and conservation of vulnerable species and habitats.

Second, our results allow to formulate recommendations concerning the targeting of future contracts. We show for instance the interest of favoring the implementation of contracts managed by public beneficiaries in a context of high land pressure, since the latter are able to bear higher opportunity costs than private owners in terms of land-value. However, private forest owners show interesting properties in terms of cost-effectiveness compared to public beneficiaries. This cost-effectiveness of private owners is not exploited under the current N2K forestry contract framework, with low levels of participation, especially from private forest owners.

This low participation can be explained by the inadequate definition of payments. Indeed, we show that the opportunity cost of timber revenues is not taken into account in the definition of payments, and this, despite the significant substitutability between

timber production and biodiversity that we demonstrate within the framework of N2K forest contracts in France. Such a definition of payments is likely to lead to the establishment of contracts with a poor level of additionality, by favoring the participation of agents characterized by low opportunity costs and by preventing the participation of those with high opportunity costs related to timber revenues.

Finally, we show that the conservation of biodiversity under the N2K forest contracts is characterized by increasing returns to scale. This suggests the establishment of more ambitious contracts allowing the conservation of multiple species and habitats, provided that these present compatible ecological requirements.

The available data did not allow us to carry out an *ex post* assessment of the cost-effectiveness of payments for N2K contracts. Such an *ex post* assessment, based on the evaluation of the direct ecological impacts of implemented conservation measures on biodiversity, could modify our results. However, the long time required to achieve conservation benefits and the stochastic nature of these conservation benefits would have invited the careful consideration of the results of such a study.

In addition, N2K contracts in forests are based on the implementation of specific conservation measures, which are defined based on knowledge of the ecology of targeted species. The effective implementation of these measures must therefore make it possible to generate the expected benefits in terms of the conservation of the targeted biodiversity. In our study, we take into account the influence of these conservation measures on the level of targeted biodiversity. Thus, the fact that our study is based on an *ex ante* approach does not diminish the interest of the results. Conducting an *ex post* evaluation is therefore not necessary to redefine more cost-effective payments for N2K contracts.

The second and third chapters of this thesis address the definition of species-specific conservation payments from a theoretical and conceptual perspective. In these chapters, we aimed at accounting for the challenges posed by biodiversity conservation which were not taken into consideration in the definition of payments for N2K contracts, especially asymmetric information and climate change.

The second chapter investigated the possibility of defining differentiated conservation payments with unobservable and heterogeneous conservation costs and benefits based on a principal-agent common-value model.

First, in this work we show the possibility of differentiating conservation contracts through a common-value model, despite the emergence of a non-responsiveness phenomenon. Second, we demonstrate the socially inefficient character of the contracts defined through a classic adverse selection model. However, it is not always cost-effective to implement contracts based on a common-value model.

The implicit consideration of spatial aspects in our model allows us to formulate recommendations on the type of conservation instruments to be implemented according to the conservation context (i.e., presence probability of the target species, additionality of effective species protection, potential of the considered landscape in terms of timber or agricultural production, etc.). We show, for example, that it is particularly efficient to base the definition of conservation contracts on a common-value model in the case of intermediate additionality of species protection. The degree of rarity of the species (i.e., species' presence probability) also favors the cost-effectiveness of contracts defined through a common-value model. However, it would be strongly inefficient to base the definition of conservation contracts on a common-value model in the case of low additionality of species protection. We also show that species-specific conservation should not rely on incentive conservation contracts at all in the case of high additionality of species protection. This theoretical work could be applied and calibrated for the implementation of specific-species conservation programs. This calibration would require information about the metapopulation dynamics of the target species, its degree of rarity and vulnerability as well as the preferences of the conservation planner in terms of additionality of species protection. Access to this latter information would be all the more important as the type of contract that can be defined significantly depends on the additionality of species protection. Our model could also be extended to multi-species conservation programs in which the ecological benefit would not be linked to the presence of a particular species but, for example, to the species richness of the property. Such an extension would require taking into account the correlation between opportunity cost and protection cost.

In the third chapter, we developed an ecological-economic model at the landscape scale and investigated the relative cost-effectiveness of various payment design options for biodiversity conservation under climate change. We consider four different payment schemes involving varying degrees of payments differentiation and targeting. Our results

show that significant cost-effectiveness gains are allowed by the implementation of targeted and differentiated payments compared to simple uniform untargeted payments. Besides, we show that classic connectivity/area trade-offs, which were highlighted in dynamic landscapes, are also observed under climate change. Finally, we show that the advantages brought by targeted payments in terms of connectivity of the conservation network are diminished by a decrease in the level of climate stability in the landscape. The conceptual nature of this work limits the direct relevance of our results in terms of conservation policy. However, we provide a first insight into the definition of payments for biodiversity conservation under climate change. In addition, the ecological model we use is a general one, which can be applied to species characterized by metapopulation dynamics. The next step would be to apply our model with real data. This would allow us to formulate finer recommendations for the considered case studies.

Future research:

It seems necessary to conduct further empirical research to provide a sound basis for a more accurate redefinition of the payments for N2K contracts. Our results stress the necessity to take into account both the “technical implementation costs” - associated with the implementation of conservation measures - as well as the opportunity costs related to foregone timber revenues in the payment definition. Moreover, the definition of cost-effective payments requires identifying and taking into account the spatial heterogeneity not only of conservation costs but also of benefits, in order to define rules for the targeting of contracts.

These two issues could be addressed in a future work through an approach combining an econometric model and species-specific ecological distribution models, such as the one carried out by Nielsen *et al.* (2017). The different types of conservation measures implemented under the N2K contracts would make it possible to link the two types of models.

As mentioned above the implementation of a conservation measure entails, on the one hand, “technical implementation costs” linked to specific management measures (e.g., removing invasive species, creation of a pond or clearing, etc.). It triggers, on the other hand, an opportunity cost linked to a change in forest management practices affecting the timber productivity of the stand. Estimating the cost of an N2K contract would

allow, on the one hand, to estimate the marginal cost associated with the implementation of the different types of conservation measures, that is, the “technical implementation cost” costs incurred. The existence of cost complementarity or substitutability between the different types of conservation measures should also be checked. On the other hand, estimating the marginal impact of a change in forest management on timber productivity would allow, based on price data, to estimate associated opportunity costs. Such an approach would thus allow to comprehensively estimate the cost of a N2K contract.

Besides, site-specific management plans identify, for each type of conservation measure, the species and habitats that benefit from it. Thus, each type of conservation measure and the associated forest management practices can be linked with its beneficiaries (species, habitats). Based on this information, the estimation of species-specific ecological distribution models accounting for the impact of pedological, climatic variables and forest management on the presence probability of target species would allow to approximate potential conservation benefits.

Accounting for economic and environmental heterogeneity in both the ecological and the econometric models would allow to identify the spatial heterogeneity of conservation costs and benefits. This would allow, on the one hand, to account for this heterogeneity in the definition of fairer payments and, on the other hand, to formulate targeting rules for the implementation of future contracts. Such targeting rules would be of significant interest for the implementation of Natura 2000 contracts which should be implemented in a prioritized manner, as suggested by the French Ministry of Environment (DGPAAT/SDDRC/C2012- 3047, MEDDE, 2012).

Besides, in the context of climate change the stochasticity of conservation benefits is likely to be exacerbated, particularly as a result of an increase in the frequency and intensity of extreme weather events. In this context, it seems optimal to share the risk due to this stochasticity of the conservation benefits between the conservation planner and the landowner by implementing result-based payments (Derissen and Quaas, 2013). In our work, we considered only input-based payments. It would therefore be interesting to extend this work and compare the relative cost-effectiveness of result-based, targeted and differentiated payments in a climate change context. Establishing such result-based payments would, in particular, allow cost savings related to monitoring in relation to targeted payments. It would be interesting to investigate whether this effect would

dominate the impact of stochasticity requiring to provide a risk premium to participants. Finally, the results of the first chapter of this thesis show the existence of increasing returns to scale within the framework of N2K contracts in forests. This shows the interest of grouping the supply of biodiversity under the same contract, for example by adopting a multi-species approach. It would be particularly interesting to extend our theoretical work by considering such a multi-species approach. As mentioned earlier, the model we develop in the second chapter of this thesis could be applied as such to a multi-species approach. However, in its current form this model does not allow to account precisely for the similarity or dissimilarity between the niche requirements of the various considered species. Taking this issue into consideration would require to develop a model integrating a cost parameter for each target species as well as the correlations between these parameters.

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Abstract: Incentive payments for biodiversity conservation – A dynamic and spatial analysis

The objective of this thesis is to study the definition of incentive payments for biodiversity conservation from an empirical and theoretical perspective. In this work, we also aim to account, in a relevant way, for spatial and dynamic ecological processes inherent to terrestrial biodiversity in the economic models that we develop. In the first chapter of this thesis, we empirically assess the cost effectiveness of incentive payments for biodiversity conservation implemented in French forests, namely Natura 2000 contracts, by undertaking an *ex ante* approach. Our results underline the inadequacy of the current definition of payments for Natura 2000 contracts and their poor calibration. This calls for a rethinking of the definition of conservation incentives. In the second and third chapter of this thesis we leave the framework of Natura 2000 contracts. We study the definition of efficient and cost-effective incentive payments in a theoretical and conceptual way, while taking into account the main challenges posed by the definition of incentive payments for biodiversity conservation. Chapter 2 explores, through a principal-agent *common-value* model, the possibility of differentiating conservation payments for private landowners when both conservation costs and benefits are heterogeneous and unobservable to the conservation planner. This chapter focuses on the impact of asymmetric information – especially of adverse selection – on the definition of payments. In Chapter 3, we investigate the impact of climate change on the definition of cost-effective incentive payments. In this chapter, we develop an integrated, dynamic and spatially explicit ecological-economic model, and study the relative cost-effectiveness of various payment design options, involving different levels of targeting and differentiation of conservation payments. The work done throughout this thesis allows us to formulate recommendations regarding the targeting and design of incentive payments for biodiversity conservation.

Keywords: Biodiversity conservation, Incentive payments, Cost-effectiveness, Asymmetric information, Natura 2000, Ecological-economic modeling, Applied econometrics.

Résumé: Paiements incitatifs pour la conservation de la biodiversité – Analyse dynamique et spatiale

L'objectif de cette thèse est d'étudier la définition de paiements incitatifs pour la conservation de la biodiversité d'un point de vue empirique et théorique. Dans ce travail, nous visons également à intégrer de façon pertinente les processus écologiques spatiaux et dynamiques inhérents à la biodiversité terrestre dans les modèles économiques que nous développons. Dans le premier chapitre de cette thèse, nous évaluons empiriquement la coût-efficacité des paiements pour contrats Natura 2000 mis en place en forêt en France par le biais d'une approche *ex ante*. Le caractère insuffisant de la définition de ces paiements et leur mauvaise calibration montre la nécessité de repenser la définition de ces dispositifs d'incitations. Dans le deuxième et troisième chapitre de cette thèse, nous étudions donc la définition de paiements incitatifs efficaces et coût-efficaces de façon théorique et conceptuelle, tout en prenant en compte les principaux enjeux posés par la définition de paiements incitatifs pour la conservation de la biodiversité terrestre. Le chapitre 2, par le biais d'un modèle principal-agent à *valeur commune*, étudie la possibilité de définir des paiements incitatifs différenciés à destinations des propriétaires privés lorsque les coûts et bénéfices de conservation sont hétérogènes et inobservables pour l'agence de conservation. Ce chapitre s'intéresse donc à l'impact de l'asymétrie d'information – en particulier du phénomène d'anti-sélection – sur la définition des paiements. Dans le chapitre 3, nous nous intéressons principalement à l'impact du changement climatique sur la définition de paiements incitatifs coût-efficaces. Dans ce chapitre, nous développons un modèle écologique-économique intégré, dynamique et spatialement explicite, nous permettant d'étudier la coût-efficacité relative de différents types de design, impliquant différents degrés de ciblage et de différenciation des paiements de conservation. Le travail réalisé dans l'ensemble de cette thèse nous permet de formuler des recommandations concernant le ciblage et le design de paiements incitatifs pour la conservation de la biodiversité.

Mots clés: Conservation de la biodiversité, Paiements incitatifs, Coût-efficacité, Asymétrie d'information, Natura 2000, Modélisation écologique-économique, Econométrie appliquée.