# La prise en compte des services écosystémiques dans l'évaluation des projets d'infrastructures de transport

Lea Tardieu

# THÈSE

Pour obtenir le grade de

Docteur

Délivré par le

**Centre international d'études supérieures en sciences agronomiques**
**Montpellier**

Préparée au sein de l'école doctorale **d'Economie et Gestion** (EDEG)
Et de l'unité mixte de recherche Laboratoire Montpelliérain d'Économie Théorique et Appliquée (UMR LAMETA)

Spécialité : **Sciences Economiques**

Présentée par **Léa TARDIEU**

**INTEGRATING ECOSYSTEM SERVICES IN THE EVALUATION OF TRANSPORT INFRASTRUCTURE PROJECTS**

**L'INTÉGRATION DES SERVICES ÉCOSYSTÉMIQUES DANS L'ÉVALUATION DES PROJETS D'INFRASTRUCTURES DE TRANSPORT**

Soutenue le 11 juillet 2014 devant le jury composé de

| | |
|---|---|
| Alain AYONG-LE KAMA, Professeur, Université Paris Ouest-Nanterre | Rapporteur |
| Serge GARCIA, Directeur de recherche, INRA | Rapporteur |
| Sophie THOYER, Professeur, Montpellier SupAgro | Examinateur |
| Bernard CHEVASSUS-AU-LOUIS, Directeur de recherche, Inspecteur général de l'agriculture | Examinateur |
| Jean-Michel SALLES, Directeur de recherche, CNRS | Directeur |
| Sébastien ROUSSEL, Maitre de conférences, Université Montpellier 3 | Co-Directeur |
| Dorothée LABARRAQUE, Chargée de mission, EGIS Environnement | Invitée |

Montpellier SupAgro n'entend donner aucune approbation ni improbation aux opinions émises dans cette thèse ; ces opinions doivent être considérées comme propres à leur auteur.

# Abstract


### Integrating ecosystem services in the evaluation of transport infrastructure projects

The purpose of this thesis is to broaden the assessment process of terrestrial transport infrastructure in the field of Ecosystem Services (ES), i.e., the benefits people derive from ecosystems. To achieve this, we first review the major challenges to integrate the ES approach into transport infrastructure decisions. This inclusion is only possible if changes in ES are explained in a spatially explicit way (Chapter 1). We illustrate this point by assessing the loss of the global climate regulation service caused by the infrastructure construction (Chapter 2). The analysis is based on the examination of a contemporary infrastructure project in Western France, and the same case study is used in the next part of this thesis. We further deepen the issue of combining direct loss of multiple ES with indirect loss due to the infrastructure impacts on landscape connectivity (Chapter 3). For both direct and indirect effects we integrate potential threshold effects on ES loss. We compare implementation options to provide an example of how choices can be improved by mapping ES loss associated with a combination of direct and indirect impacts. Finally, we provide a test of the usefulness of the ES consideration into environmental impact assessment and cost-benefit analysis in order to assess the additional information it may bring (Chapter 4). We show that this analysis can provide guidance at different stages of transport project: from the preliminary studies to the study of the final implementation option. For environmental impact assessment, the consideration of ES opens the possibility of measuring ES loss providing a means for selecting among a set of route options for the infrastructure. For cost-benefit analysis, since ES loss induced by the selected route is expressed in monetary terms, it can be integrated as a standard social cost in the analysis, allowing a more efficient control of natural capital loss. As a result, this may help project stakeholders to better consider the effects of the infrastructure implementation.



**Keywords:** Ecosystem services, Terrestrial transport infrastructures, Environmental impact assessment, Cost-benefit analysis, Economic valuation, Spatial assessment.

## Résumé

### L'intégration des services écosystémiques dans l'évaluation des projets d'infrastructures de transport

L'objectif de cette thèse est d'intégrer la notion de Services Écosystémiques (SE), i.e., les bénéfices que la société retire du fonctionnement des écosystèmes, dans le cadre de l'évaluation des projets d'infrastructures de transports terrestres. Pour cela, nous commençons par mettre en lumière les différents défis associés à l'intégration des SE dans les décisions d'implantation d'infrastructures de transport. L'intégration ne peut être réalisée que si l'estimation de la perte de SE est faite de manière spatialement explicite (Chapitre 1). Puis, nous illustrons ce point à travers l'étude de la perte d'un service : la régulation du climat global (Chapitre 2). L'analyse est basée sur l'examen d'un projet d'infrastructure contemporain dans l'ouest de la France, et le même cas d'étude est utilisé dans la suite de cette thèse. Nous approfondissons ensuite la question de la combinaison de la perte directe et de la perte indirecte de SE due aux impacts de l'infrastructure sur la connectivité des entités spatiales (Chapitre 3). Pour les deux types d'impacts, nous intégrons des seuils potentiels sur la fourniture de services en proposant une méthode de prise en compte pour des écosystèmes particulièrement sensibles. Nous comparons différentes options de tracé afin de donner un exemple de la manière dont les choix pourraient être améliorés en cartographiant les pertes directe et indirecte de SE. Enfin, nous montrons l'intérêt de la prise en compte des SE dans l'étude d'impact environnemental et le bilan socio-économique de manière à mesurer l'information supplémentaire qu'apporte une telle intégration (Chapitre 4). Nous montrons que ce type d'analyse peut orienter différentes étapes d'un projet d'infrastructure, des études préliminaires jusqu'à l'étude du tracé final. Dans le cas des études d'impact environnemental, l'intégration de ces considérations permet de mesurer la perte de services engendrée par chaque tracé d'infrastructure et d'intégrer ces pertes en tant que nouveau critère de choix de tracé. Concernant le bilan socio-économique, la perte de services exprimée en termes monétaires permet de donner des informations quant à la perte sociale engendrée par le tracé final. Ceci peut aider les parties prenantes des projets à mieux appréhender les effets engendrés par la réalisation de l'infrastructure.



**Mots clés :** Services écosystémiques, Infrastructures de transport terrestres, Étude d'impact environnemental, Analyse coût-avantage, Evaluation Economique, Evaluation spatiale.

# Remerciements

L'aboutissement de ce travail de thèse n'aurait pas été possible sans le soutien apporté par un grand nombre de personnes, et pour cette raison je tiens à les remercier.

Avant tout, je voudrais remercier EGIS Environnement pour avoir financé cette thèse. Qu'ils trouvent ici toute ma gratitude pour la confiance qu'ils m'ont accordée en me confiant ce sujet. Je voudrais remercier plus particulièrement Dorothée Labarraque, pour sa disponibilité, son implication, sa ténacité et pour l'enthousiasme dont elle a fait preuve dès le départ pour ce travail. Une telle motivation a été d'une grande aide pour garder le cap. Merci à Bénédicte Authié, pour sa gentillesse, sa disponibilité et ses talents de cartographe. Merci aussi à Virginie Defromont et plus globalement, merci à toute l'équipe d'Egis structures et environnement pour leur accueil chaleureux lors de mes virées à Toulouse.

Je remercie Alain Ayong Le Kama et Serge Garcia qui ont accepté d'être les rapporteurs de cette thèse. Je remercie également Sophie Thoyer et Bernard Chevassus-au-Louis qui ont accepté d'être examinateurs.

Merci à mes directeurs de thèse Sébastien Roussel et Jean-Michel Salles, pour la confiance qu'ils m'ont accordée depuis le stage de master. Plus particulièrement, merci à Sébastien pour son soutien, sa patience, et l'aide qu'il m'a apportée tout au long de ce travail. Merci à Jean-Michel d'avoir accepté de diriger ce travail, pour ses conseils et ses encouragements.

Mes remerciements vont aussi à John Thompson pour les discussions constructives que l'on a pu avoir, pour sa gentillesse, sa disponibilité, son efficacité, et pour s'être intéressé et impliqué dans ce travail. Son soutien a été d'une importance primordiale. Merci à Tina Rambonilaza, Mohamed Hilal, Luc Baumstark, Harold Levrel d'avoir accepté de participer à mon comité de thèse et m'avoir aiguillé dans un moment de doute. Merci également à ceux qui nous ont permis de communiquer sur ce travail: B. Burkhard, N. Crossman, S. Nedkov, K. Petz et R. Alkemade, P.A. Mahieu, B. Kriström, et P. Riera. Merci aux referees anonymes (que je ne citerai donc pas) qui nous ont permis d'améliorer ce travail grâce à des conseils constructifs et avisés. Merci aux membres du CGDD et plus particulièrement à P. Puydarrieux et Y. Autret pour leurs invitations à communiquer ce travail et pour leur intérêt. Merci également aux membres du LEF pour leur séminaire enrichissant, leur accueil et leur bonne humeur. Merci à l'unité InfoSol de nous avoir fourni des données précieuses.

Merci aux membres du LAMETA pour m'avoir permis de réaliser cette thèse dans de bonnes conditions, notamment au niveau documentaire -aparté pour Laurent Garnier, merci pour ton efficacité et ton entrain!-, et pour leur accueil. Cette expérience de vie au laboratoire aura été très enrichissante et instructive pour moi. Merci également de m'avoir donné l'opportunité de participer à des colloques qui ont été sources d'inspiration. Enfin un grand merci aux relecteurs attentifs (ceux qui n'étaient pas obligés) : Nicolas Quérou, Laure Kuhfuss, John Thompson, Alexis Bédiée, François Tardieu, et Christine Granier (j'ai presque eu l'impression que ce travail t'avait intéressé même si ça ne parle ni de foot, ni de la feuille 6 d'*Arabidopsis Thaliana*).

Et je tiens enfin à remercier ceux qui ont contribué à rendre ma vie douce au quotidien, c'est à dire :

- mes amis-collègues et collègues-amis (je ne sais plus), bref compagnons de thèse. Votre soutien au quotidien, pendant ces trois ans et demi, a été d'une grande aide pour mener à bien ce travail. Julia, la thèse n'aurait pas été la thèse sans toi (et sans tes théories). Je crois qu'on a muri ensemble, et ça c'est beau ! Ceci s'applique également à Laure (la chanceuse), Mathieu (le comique), et Sophie (la maman). Á celles qui sont parties Estelle, Agathe et à ceux qui sont arrivés Xavier, Aziz, Philippe, Guillaume L.B. et Nicolas (tu noteras que tu es placé parmi les jeunes en guise de remerciement - cf. plus haut).
- mes amis pas-collègues-mais-amis-tout-de-même qui m'ont aidé à rester connectée à la « vraie vie » (ou du moins qui ont essayé): Fanny, Lorène, Sarah,(Générale) Carole, Jimmy, Anthony, Camille, Kevin et Jeff. Mais aussi mes frères et soeurs Antoine, Popi, Thomas, Émilie et Noémie.
- mon papa qui m'a poussé à persévérer, pour son aide, ma maman qui m'a encouragé, mes beaux-parents, mes beaux grands-parents, mes grands beaux-parents et beaux beaux-parents. Pour vos soutiens, pour m'avoir aidé à moins douter, pour toutes les fois où vous m'avez changé les idées, pour les bons repas (parce que ça compte tout de même), et pour bien d'autres choses encore.
- Alexis, pour ton soutien sans failles, tes encouragements au quotidien, mais aussi pour tes talents de "personal assistant", coach moral, clown, scribe, bref pour être toi, et pour avoir été là.

Et ma dernière pensée va à Guillaume Bédiée pour que ton nom reste inscrit ici.

# Contents



[1]This work has been published as: Tardieu, L., Roussel, S., Salles, J.-M., 2013. Assessing and mapping global climate regulation service loss induced by terrestrial transport infrastructure construction. Ecosystem Services 4, 73-81.

[2] A preliminary version of this work has been accepted for publication in Revue d'Économie Politique. The full version of the paper has been submitted as: Tardieu, L., Roussel, S., Thompson, J. D., Labarraque, D., Salles, J. M. Combining direct and indirect impacts on ecosystem service loss associated with infrastructure construction.

# List of Figures

# List of Tables

# List of abbreviations

ARIES: ARtificial Intellingence for Ecosystem Services

CBA: Cost Benefit Analysis

CBD: Convention on Biological Diversity

CGDD: Commissariat Général au Développement Durable

CLC: Corine Land Cover

COPI: Cost Of Policy Inaction

EIA: Environmental Impact Assessment

EFESE: Évaluation Française des Écosystèmes et des Services Écosystémiques

ES: Ecosystem Services

GDP: Gross Domestic Product

GIS: Geographical Information System

IFEN: Institut Français de l'Environnement

InVEST: Integrated Valuation of Environmental Services and Tradeoffs

IPBES: Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

IPCC: Intergovernmental Panel on Climate Change

LULC: Land-Use/Land-Cover

MEA: Millenium Ecosystem Assessment

NPV: Net Present Value

TEEB: The Economics of Ecosystems and Biodiversity

SNB: Stratégie Nationale pour la Biodiversité

SETRA: Service d'Étude sur le Transports, Les Routes et leurs Aménagements

SOC: Soil Organic Carbon

TTI: Terrestrial Transport Infrastructure

UKNEA: United Kingdom National Ecosystem Assessment

UNEP: United Nation Environment Program

VCS: Vegetation Carbon Stocks

WTP: Willingness To Pay

# Introduction Générale

## La prise en compte des externalités environnementales dans les projets d'infrastructure de transport : Une nécessité grandissante

Les infrastructures de transports terrestres représentent 39% des surfaces artificialisées en France (IFEN, 2006). Le réseau ferré s'étend sur plus de vingt-neuf mille kilomètres de lignes et le réseau routier représente plus de 1 million de kilomètres en 2012 (CGDD, 2014). Ces réseaux incluent également d'importantes infrastructures complémentaires, comme par exemple des équipements de sécurité, des systèmes de caténaires, ou encore des stations électriques pour le rail. Les infrastructures de transport jouent un rôle essentiel dans le développement économique des nations (représentant 4.7% du PIB et 5.8% du total des salariés selon l'INSEE). Elles constituent un facteur de liberté contribuant au bien-être des populations (gain de temps, désenclavement du territoire, etc.). Elles ont donc une réelle utilité sociale. Cependant, la mise en place de ces infrastructures s'accompagne inévitablement d'effets indésirables sur la biodiversité et les écosystèmes. Ces pressions sont de trois types :

- La perte d'habitat ou la réduction du domaine vital des espèces sauvages due à l'artificialisation des sols ;
- La fragmentation des habitats : soit l'isolement des patchs d'habitats ou morcèlement du paysage affectant sa connectivité ;
- La dégradation de la qualité des habitats par des nuisances sonores ou visuelles, des vibrations, la salinisation ou encore les pollutions engendrées par la construction des infrastructures et le trafic qu'elles permettent ensuite.

La France s'est engagée à limiter ce type de pressions. Elle est signataire de la Convention sur la Diversité Biologique (CDB) au cours de la Conférence des Nations Unies sur l'Environnement et le Développement de Rio de Janeiro en 1992. Conformément à cet engagement international, la France s'est dotée d'une Stratégie Nationale pour la Biodiversité (SNB) en Février 2004, révisée ensuite en 2009. La SNB est structurée selon quatre grandes orientations visant à mobiliser tous les acteurs, reconnaître sa valeur au vivant, améliorer la prise en compte par les politiques publiques et développer la connaissance scientifique concernant la biodiversité. Ces orientations sont déclinées en plans d'action sectoriels dont celui concernant les infrastructures de transports terrestres.

Il devient donc essentiel d'analyser la manière de mieux prendre en compte l'environnement dans les projets d'infrastructures pour respecter la SNB, ce qui conduit à distinguer deux voies.

- La *première est l'étude d'impact environnemental* visant à déterminer le meilleur tracé d'infrastructure au sens de la minimisation de l'impact sur différents biens environnementaux (milieux naturels, ressources en eau, sylviculture et agriculture, et paysage). La recherche de ce tracé se fait dans le but de (1) éviter autant que faire se peut les impacts et les dégradations environnementales des projets, (2) réduire les effets ne pouvant être évités en adaptant les caractéristiques du projet, et (3) compenser les impacts résiduels dans le but de générer des avantages écologiques au moins équivalents à la perte.

- La *seconde consiste à prendre en compte les externalités environnementales*, traduites en termes monétaires, dans l'évaluation socio-économique de l'infrastructure. S'agissant d'investissements publics, l'évaluation ne peut se limiter à un bilan financier des impacts des infrastructures et devrait prendre en compte un ensemble d'effets ne faisant pas l'objet d'une régulation marchande : les externalités. Les externalités sont définies comme des effets, positifs ou négatifs, engendrés par l'action d'un agent économique sur d'autres agents ne l'ayant pas choisi, sans qu'il n'y ait de compensation monétaire en contrepartie. En présence d'externalités, les décisions individuelles ne peuvent conduire à l'optimum social puisqu'elles reposent sur un calcul ne prenant en compte que les coûts privés et non l'intégralité des coûts que la société devra supporter. Le marché n'est donc pas en mesure de corriger les dysfonctionnements causé par l'externalité, ni de réguler les phénomènes de dégradation. Si aucune régulation n'est mise en place, alors les usages marchands dominent les usages non marchands comme c'est souvent le cas pour les ressources naturelles, conduisant à une surexploitation de celles-ci. Le rapport Boiteux II (2001) a marqué une première avancée concernant la prise en compte d'externalités environnementales dans le bilan socio-économique des projets d'infrastructures. Dans ce rapport, des valeurs tutélaires ont été définies pour trois types de nuisances engendrées par le fonctionnement des infrastructures de transport : la pollution de l'air au niveau local, la pollution de l'air au niveau global et les nuisances sonores. La prise en compte de ces externalités a ensuite été inscrite dans l'instruction cadre relative aux grands projets d'infrastructure, rendant règlementaire leur inclusion dans les bilans.

La révision de la SNB effectuée en 2009 visait à renforcer les objectifs prévus en 2004 mais également à intégrer des préoccupations émergentes comme le changement climatique et les services rendus par les écosystèmes à savoir les services écosystémiques.

## La notion de service écosystémique permet de communiquer sur la dépendance de nos sociétés au système de support écologique, mais reste encore peu utilisée dans la pratique.

La notion de services écosystémiques, introduite en 1981 par Ehrlich et Ehrlich, se définit comme les avantages retirés par les populations humaines du fonctionnement des écosystèmes. La définition et la typologie des services sont variables (Costanza, 2008). Afin de clarifier un débat souvent confus, le Millenium Ecosystem Assessment (MEA, 2005) a proposé de les classer en quatre grandes catégories :

- Les services de support, non directement utilisés par l'homme mais nécessaires à la durabilité d'un système (formation des sols, développement du cycle nutritionnel, maintien de la biodiversité, etc.) ;

- Les services de prélèvement, qui conduisent à des biens consommables comme la nourriture, les matériaux et fibres, l'eau douce ou le bois de feu. Ces services font l'objet d'un usage direct ;

- Les services de régulation, définis comme les bénéfices retirés de la régulation des processus écosystèmiques (maintien de l'humidité relative de l'air, de la qualité de l'eau, contrôle des maladies des plantes), ces services font l'objet d'un usage indirect ;

- Les services culturels tels que les bénéfices retirés de la récréation de plein air correspondant à des avantages non matériels, faisant l'objet d'un usage direct sans entrainer leur consommation.

Le concept de services écosystémiques est désormais perçu comme une façon de démontrer explicitement l'existence d'interactions et d'interdépendances entre les sociétés humaines et l'environnement naturel. La popularisation de ce concept a été principalement initiée par le MEA (2005), les études menées par le TEEB (2010), ou encore le rapport Chevassus-au-Louis et al (2009) en France. Le principal résultat de ces études a été de mettre en évidence le fait qu'une large proportion des écosystèmes a subi et continue à subir de graves dégradations, affectant leur capacité à fournir des services bénéficiant aux populations, alors même que la demande de ces services est en augmentation. On perçoit ainsi plus clairement que la préservation des écosystèmes naturels n'est pas une gêne par rapport au bien-être des citoyens, mais qu'elle constitue un capital naturel, un actif important pour nos sociétés (Liu et al, 2010).

Pourquoi évaluer économiquement les services et donner une valeur à des objets le plus souvent considérés comme n'en ayant pas ? La réponse consensuelle est que les services rendus par les écosystèmes sont principalement des actifs non marchands n'ayant par conséquent pas de prix. Cependant, une logique purement financière conduit à agir comme s'ils n'avaient pas de valeur, aboutissant à une mauvaise allocation des ressources. Des recherches ont donc été consacrées à la classification des services, leur quantification et l'évaluation économique des services écosystémiques.

L'objectif premier de l'évaluation économique des services écosystémiques est d'apporter des éléments d'information, les plus proches possibles de la réalité, dans le but d'appuyer les décisions publiques ou privées impliquant l'environnement. Outre l'intégration des services dans les évaluations de politiques publiques au même titre que d'autres enjeux de société, l'évaluation peut également être faite dans le but de :

- Définir les stratégies de conservation et en éclairer les priorités. C'est le rôle de l'analyse coût-efficacité qui vise à répondre à la question : où un euro investi dans la conservation serait le mieux employé ?

- Fixer le niveau d'effort consenti pour la conservation. Est-il plus pertinent de détruire tel écosystème ou de renoncer à tel projet qui les menace ?

- Communiquer sur l'ordre de grandeur des enjeux globaux dans des termes susceptibles de favoriser la prise de conscience de leur importance par les décideurs publics et le public en général. Ceci pourrait motiver ainsi des stratégies ambitieuses en termes de conservation.

De façon générale, les évaluations sont donc mises en oeuvre pour motiver et justifier l'action collective.

Peu d'applications et de recherches ont été menées concernant l'impact des infrastructures sur les services écosystémiques depuis le rapport Chevassus-au-Louis et al (2009) et la révision de la SNB, alors même que la réflexion était clairement engagée. En effet, peu d'exemple d'intégration en France comme à l'international, peuvent être cités (Broekx et al, 2013 ; DEFRA, 2008 ; SETRA, 2010). Par ailleurs, les études stagnent souvent au stade de l'encouragement à la réflexion, comme cela a été récemment le cas dans le dernier rapport Quinet (2013). D'autres travaux enfin proposent de monétariser les dégradations faites à environnement engendrées par les projets d'infrastructures comme dans le cas de l'étude de l'Office fédérale du développement territorial de Suisse (2003) ou encore le Projet ExternE basé sur le Rapport INFRAS / IWW (2004). Ces rapports ne traitent cependant pas explicitement de la perte de services mais de l'évaluation de la perte d'habitats. L'évaluation est faite à partir de coûts de restauration et de coûts de remplacement pour la perte d'habitats. La fragmentation est quant à elle approchée par des coûts de remplacements relatifs à la mise en place de passages à faune. Ces valeurs ne reflètent pas d'après nous de manière exhaustive la perte subie par la société de la dégradation des écosystèmes, puisqu'elles ignorent les flux de services.

## Stratégie pour une prise en compte des services écosystémiques dans les projets d'infrastructures de transport

Comment répondre à une demande de mobilité croissante tout en cherchant à limiter le rythme d'érosion des services rendus par les écosystèmes ?

Nous avons adopté une logique *ex-ante* consistant à intervenir au niveau du projet. La prise en compte des services écosystémiques à ce niveau permettrait tout d'abord d'élargir le champ des impacts pris en compte dans les études environnementales actuelles pour permettre de dépasser le stade des obligations règlementaires. Ceci permettrait de prendre en compte les effets de l'infrastructure à une échelle plus large et de mesurer les pertes socioéconomiques engendrées. La motivation essentielle de ce type d'intégration est donc de définir les objectifs à atteindre dans la recherche du meilleur tracé d'infrastructures, et éventuellement d'apporter des éléments d'information, exprimés quantitativement, concernant les externalités relatives à la perte de services liées à l'implémentation de celle-ci.

Cette tâche nécessite de mobiliser différentes disciplines, notamment l'écologie, l'économie, et la géographie. En effet, incorporer les coûts associés aux changements d'utilisation des terres requiert la quantification de ces changements en termes physique et économique. Par ailleurs, la fourniture de services, leur demande et les avantages qui y sont associés dépendent de nombreux facteurs variant spatialement (comme les conditions climatiques, la topographie, la distance par rapport aux bénéficiaires, ou le nombre d'écosystèmes produisant un service similaire aux alentours). L'ampleur de l'impact d'une infrastructure et la perte de services engendrée par sa construction dépend de la manière dont ces facteurs ont été modifiés dans l'espace. L'analyse devrait enfin être construite de manière à apporter un cadre méthodologique pratique, reproductible et adapté au contexte de ces projets mais également à produire une méthodologie mobilisant autant que faire se peut les outils et les données déjà rendues disponibles par les analyses existantes.

Nous arrivons ainsi à des questions de recherche plus précises : **Dans quelle mesure peut-on cartographier la perte de services écosystémiques induite par les options de tracés d'une infrastructure de transport ? Peut-on en élaborer un nouveau critère de choix de tracé ? Et dans quelle mesure ce critère peut-il s'intégrer aux outils d'évaluation existants que sont l'étude d'impact environnemental et le bilan socio-économique ?**

La thèse est structurée en quatre chapitres (écrits sous la forme d'articles). Le Chapitre 1 est une revue de la littérature visant à définir les différents défis associés à la représentation spatiale des services écosystémiques. Il dresse un état de l'art des pratiques en matière de spatialisation des services. Ces éléments de cadrage étant fournis, de nouvelles questions relatives à l'évaluation spatiale de la perte de services écosystémiques dans le cas des infrastructures de transport sont soulevées et détaillées. Le Chapitre 2 et le Chapitre 3 sont essentiellement méthodologiques : le premier cherche à tester la prise en compte des services écosystémiques dans un cas simple (un seul service), le second dans des cas plus complexes (de multiples services devant intégrer différents facteurs spatiaux). Ces deux chapitres ont fait l'objet de publications dans des revues à comité de lecture. Enfin, le Chapitre 4 définit la manière dont les services peuvent être intégrés de manière adaptée au cadre d'évaluation actuel, et montre l'intérêt d'une telle approche à différents stades de l'évaluation d'un projet d'infrastructure [3].

[3]Cette thèse a été financée par EGIS structures et environnement (filiale de la Caisse des Dépôts et Consignation). Elle a fait l'objet de la production d'un guide méthodologique (non joint à la thèse car il est maintenant propriété d'EGIS). Le guide donne un cadre méthodologique et retrace les méthodes développées de manière à ce qu'elles puissent être reproduites sur plusieurs projets d'infrastructures (lignes à grande vitesse et autoroutes).

Les apports de cette thèse peuvent ainsi être synthétisés de la manière suivante :

- Contribuer à clarifier les questionnements pouvant émerger de la prise en compte des services écosystémiques dans les projets d'infrastructures de transport, sujet peu investi à ce jour ;

- Fournir un cadre méthodologique pour l'analyse des impacts des infrastructures sur la fourniture de différents services fournis par différents écosystèmes ;

- Appliquer ce cadre méthodologique à un projet afin d'illustrer les informations économique et environnementales que peut apporter ce type d'analyse. Pour ce faire nous intégrons la perte de services écosystémiques dans l'étude d'impact environnementale et le bilan socio-économique d'un projet réel (voir Encadré 1 ci-dessous). Ce travail est donc un travail de recherche appliquée visant à fournir des outils méthodologiques ou du moins à avancer dans ce sens.

En revanche, la thèse ne traite pas :

- Des valeurs de non usage retirées des écosystèmes (existence, héritage, procuration) : nous avons en effet considéré que les estimations sont trop peu nombreuses et nous n'en comprenons pas encore tous les effets. Nous avons préféré nous concentrer sur les valeurs d'usage ;

- De la compensation écologique. Il s'agit là d'un sujet vaste, impliquant un nombre important de questions supplémentaires, qui mériterait à son tour d'être investi dans un travail de recherche spécifique ;

- Du débat entourant l'évaluation économique des services écosystémiques et la mise en place d'analyse coût-avantage. Nous en approchons dans l'exposé des limites de notre travail, mais n'approfondissons pas ce débat car beaucoup l'ont déjà fait, et il est probable que nous n'y apporterions rien de plus.

# Références


Boiteux (Rapport du groupe d'experts présidé par), 2001. Transport : choix des investissements et coûts des nuisances. Commissariat Général du Plan, Paris.

Broekx, S., Liekens, I., Peelaerts, W., De Nocker, L., Landuyt, D., Staes, J., Meire, P., Schaafsma, M., Van Reeth, W., Van den Kerckhove, O., Cerulus, T., 2013. A web application to support the quantification and valuation of ecosystem services. Environmental Impact Assessment Review 40, 65-74.

DEFRA. Case study to develop tools andmethodologies to deliver an ecosystems approach-Heysham to M6 link DEFRA research project nr0110. (Online) Available from : http://randd.defra.gov.uk/Document.aspx?Document=NR0110_7329_FRA.pdf2007.

Chevassus-au-Louis, B., Salles, J.M., Pujol, J.L., Bielsa, S., Martin, G., Richard, D., 2009. Approche économique de la biodiversité et des services liés aux écosystèmes : contribution à la décision publique. Documentation française, Paris.

Commisariat Général au Dévelopement Durable, 2014. Chiffres clés du transport- édition 2014, in : Service de l'observation et des statistiques, Repères, Paris. (Online) Available from : http://www.statistiques.developpement-durable.gouv.fr

Costanza R., 2008. Ecosystem services : Multiple classification systems are needed. Biological Conservation 141, 350-352.

Ehrlich, P.R., Ehrlich, A.H., 1981. Extinction : the causes and consequences of the disappearance of species. Random House New York.

Géniaux, G., 1999. évaluation et régulation des impacts environnementaux. Thèse, GREQAM, Marseille.

Institut Français de l'Environnement (IFEN), 2006. L'environnement en France. MEDDE, Paris. (Online) Available from :
http://www.statistiques.developpement-durable.gouv.fr/publications/

SNB, (Online) Available from :
http://www.developpement-durable.gouv.fr/-La-Strategie-nationale-pour-la-.html

Liu, S., Costanza, R., Farber, S., Troy, A., 2010. Valuing ecosystem services. Annals of the New York Academy of Sciences 1185, 54-78.

Millenium Ecosystem Assessment, 2005. Ecosystem and well-being : A framework for assessment. Island Press Blurb, Washington DC.

Office Fédérale du Développement Territorial de Suisse, 2003. Externe kosten des verkehrs im bereich natur und landschaft Econcept, Nateco. .

Quinet, E., Baumstark, L., Bonnet, J., Croq, A., Ducos, G., Meunier, D., Rigard-Cerison, A., Roquigny, Q., 2013. Evaluation socioéconomique des investissment publics in : Française, L.d. (Ed.), Rapports et documents. Commissariat Général à la Stratégie et à la prospective, Paris.

INFRAS/IWW., 2004. External costs of transport : update study. . Schreyer, C., Maibach, M., Rothengatter, W., Doll, C., Schneider, C., Schmedding, D. The International Union of Railways (UIC) (Ed.), Karlsruhe/Zürich/Paris.

SETRA, 2010. Monétarisation des externalités environnementales. Rapport d'études, MEEDDM. Paris.

TEEB, 2010. The Economics of Ecosystems and Biodiversity : Ecological and Economic Foundations. In : Kumar, P. (Ed.), Earthscan, London, Washington.

# General Introduction

## The consideration of environmental externalities in infrastructure projects: a growing necessity

Transport infrastructures represent 39% of converted areas in France (IFEN, 2006). The railroad network extends over more than twenty nine thousand kilometers of lines and the road network represents more than 1 million kilometers in 2012 (CGDD, 2014). These networks also include additional infrastructures, as for instance safety equipment, catenaries systems, or electrified rails. Transport infrastructures play an essential role in economic development, namely 4.7 % of the GDP and 5.8 % of the total employees according to INSEE. They are associated with a need for freedom contributing to population well-being (*e.g.* time saving, opening up of the territory). Hence, they have a clear social utility. However, they also inevitably involve pressures on biodiversity and ecosystems, namely:

- Species habitat loss or the reduction of species vital domain due to the conversion of natural areas;
- Habitat fragmentation: habitat patches isolation or landscape division affecting its connectivity ;
- Threatening of habitats quality due to noise or visual disturbance, vibrations, salinization, or pollution induced by the construction and further by traffic.

France has committed to limit these pressures. It has signed the Convention on Biological Diversity (CBD) since the Conference of United Nations on the Environment and the Development of Rio de Janeiro in 1992. According to this international commitment, France has set up a National Strategy for the Biodiversity (NSB) in February 2004, which was then revised in 2009. The SNB is structured according to four key orientations to mobilize all actors, recognize biodiversity value, improve the consideration by public policies, and develop the scientific knowledge related to biodiversity. These orientations guide sectorial action plans including a plan related to transport infrastructures.

Hence, it becomes essential to analyse the way to integrate environmental considerations into transport infrastructure projects to comply with the NSB, which leads in distinguishing two ways.

- The first is *the Environmental Impact Assessment (EIA)*, which aims at determining the best implementation options for infrastructures in respect to environmental goods such as natural areas, water resources, agricultural areas and forestry or landscape scenic beauty. Implementation options have to be designed in order to (1) avoid, as far as possible, regulated protected areas; (2) to mitigate residual impacts by adapting the project characteristics in order to correct identified damages, and finally (3) to compensate residual impacts in order to generate ecological benefits by promoting favourable environmental actions such as species habitats creation or ecosystem restoration, intended to be at least equivalent to the loss incurred.

- The second consists in *the integration of environmental externalities* in the cost-benefit analysis of the infrastructure. This assessment should not be limited to a financial balance of the infrastructure impacts and has to take into account a series of effects that are not controlled by the market, usually called external effects or externalities. Externalities are defined as the positive or negative effects caused by an economic agent on other agents who did not choose it, with no monetary compensation in return. In presence of externalities, individual decisions cannot lead to a social optimum because they are based on calculations that only include private costs, and not the complete collective costs that society will incur. The market is therefore unable to correct the dysfunctions caused by the externality, nor to regulate the degradation phenomena. If no regulation is inserted, market uses override the non-market uses. This is most often the case for natural resources, leading to their overexploitation. The Boiteux II report (2001) has given a step forward for better taking into account environmental externalities in the cost-benefit analysis of infrastructure projects. In this report, reference values were defined and can be integrated in public economic calculations, translating the effects of local/global pollution and noise disturbance caused by transport infrastructures. Taking these effects into account in the cost benefit analysis became mandatory since the Framework Instruction (2005).

The SNB revision in 2009 aimed at strengthening the 2004 objectives, but also to integrate emerging concerns as climate change and ecosystem services.

## The ecosystem service concept allows to communicate on human societies' dependence on the ecological support system, but is still little used in practice.

The concept of ecosystem services (Ehrlich and Ehrlich 1981) is defined as the benefits that human population derives from natural ecosystems. The definition and typology of services vary across studies (Costanza, 2008). To clarify an often confusing debate, the Millennium Ecosystem Assessment (MEA, 2005) has proposed four categories of services:

- Supporting services, not directly used by humans but necessary to the system durability (soil formation, nutrient cycle, biodiversity maintenance, etc.);

- Provisioning services, leading to consumption goods such as food, raw materials, fibers, freshwater or firewood. These services are directly used by humans;

- Regulating services, defined as the benefits derived from the regulation of ecosystem processes (local climate regulation, water quality, biological control, etc). These services are used indirectly.

- Cultural services defined as the benefits derived from outdoor recreation or landscape scenic beauty, corresponding to non-materials benefits, directly used without causing their consumption.

Ecosystem services are now perceived as a way to explicitly demonstrate the existence of interactions and interdependencies between human societies and natural environment. The promotion of the concept was initiated by the MEA (2005), the TEEB studies (2010), or the Chevassus-au-Louis et al report (2009) in France. The result of these studies has been to highlight the fact that a large proportion of ecosystems has suffered and continues to suffer serious damage, thereby affecting their capacity to supply increasingly demanded services. It is now widely perceived that ecosystem preservation does not result in sacrifices of our well-being, but is a preservation of a natural capital that is essential for the society.

How can one assign a monetary value to goods usually considered as priceless, thereby valuing services in economic studies? The conventional view is that ecosystem services are non-market assets so they are priceless. However, a purely financial logic leads to act as if they had no value, leading to resource mis-allocation. Research on ecosystem services has therefore attracted attention in recent years to improve the ways to classify, quantify or value services in order to assess the effect of land-use changes. The primary objective of the valuation of ecosystem services is to provide information supporting public and private decisions. In addition to integrate ecosystem services in assessments of public policies as well as other social issues, the economic valuation can also aim to:

- Define conservation strategies and highlight priorities in cost-effectiveness analyses aiming at answering the question: “what is the best use of one euro invested in conservation?”
- Fix the level of conservation effort. Is it more appropriate to destroy one ecosystem or to waive the project that threatens it?
- Communicate on the magnitude of global issues in terms that promote awareness of their importance by policy makers and the general public, and motivating ambitious conservation strategies.

Overall, assessments are made to motivate and justify collective actions.

Few real-size studies and theoretical research have been conducted to assess the infrastructure impacts on ecosystem services since the report of Chevassus-au-Louis et al (2009) and the revision of the SNB report, even though a momentum was given. Indeed, few examples of integration, in France and abroad, can be cited (Broekx et al, 2013; DEFRA, 2008; SETRA, 2010). In addition, studies often stagnate at the stage of encouraging thinking, as in the last Quinet report (2013). Other studies propose to monetize environmental degradation generated by infrastructure projects as in the study of the Federal Office for Spatial Development in Switzerland (2003) and the project "Internalisation Measures and Policies for All external Cost of Transport (IMPACT) based on the INFRAS / IWW Report (2004). However, these reports do not explicitly address the loss of services, but the assessment of habitat loss, in such a way that the valuation only includes restoration costs and replacement costs for habitat loss. In these studies, fragmentation impacts are approximated by replacement costs associated to fauna passages. I propose that these values do not reflect the overall loss incurred by society when an ecosystem degradation occurs, because it ignores ecosystem services flows.

## Strategy for a consideration of ecosystem services in transport infrastructure projects

How the growing demand for mobility can be satisfied while limiting the erosion rate of ecosystem services?

We have adopted in this thesis an *ex-ante* approach that consists in intervening at the project level. The consideration of ecosystem services as such can first help to broaden the scope of impacts in current environmental studies, beyond regulatory requirements. This would allow taking into account the infrastructure effects at the wide scale, as the socio-economic losses caused. The main motivation for this type of integration is to define the objectives to reach in finding the best implementation option, and possibly to provide information elements expressed quantitatively, on externalities related to the ecosystem services loss.

This thesis mobilizes various disciplines, including ecology, economics and geography. Incorporating the costs associated with land use changes into transport infrastructure has required the quantification of these changes in physical and economic terms. In addition, the services supply, demand and benefits were considered as depending on factors varying spatially (such as climatic conditions, topography, beneficiaries' location, the positions of ecosystems producing a similar service in the same area). Hence, we have evaluated the magnitude of the impact of a given infrastructure on services, taking into account the way in which these factors are spatially affected. Finally, it has been considered that the proposed analysis should result in a methodological framework that is reproducible in spite of different project contexts by mobilizing tools and information available in existing analysis.

Overall, the precise research questions addressed in this thesis are: **How can one assess and map ecosystem services loss caused by different options for a transport infrastructure? Can one develop a new selection criterion? To what extent can this criterion be integrated in existing assessment tools such as environmental impacts and cost benefit analyses?**

The thesis is structured in four chapters, written as papers of academic journals. Chapter 1 is a literature review that defines the challenges associated with the spatial representation of ecosystem services, and provides a state-of-the-art of current mapping techniques.

Issues related to the spatial assessment of ecosystem services loss in the case of transport infrastructure project are then identified and detailed. Chapter 2 and Chapter 3 are essentially methodological. Chapter 2 tests the inclusion of ecosystem services in a simple case with one service. Chapter 3 addresses more complex cases with multiple services that involve different spatial factors. These two chapters have been published in peer reviewed journals. Finally, Chapter 4 defines how the ecosystem services can be integrated to the current assessment framework, and shows the interest of such an approach at different stages of the assessment of an infrastructure project [4].

The contributions of this thesis can be summarized as follows:
- Contribute to clarify issues that arise from the inclusion of ecosystem services in transport infrastructure projects, a subject that has received scarce interest until now.
- Provide a methodological framework for analyzing the infrastructure impacts on multiple services delivered by different ecosystems;
- Apply this methodological framework in a case study, in order to illustrate the economic and environmental information that can provide this type of analysis. We have integrated the loss of ecosystem services in the environmental impact assessment and the cost benefit analysis. Hence, this work is an applied research that still aims at providing methodological tools.

Conversely, this thesis does not address:
- Services non-use values (existence values, inheritance): We have indeed considered that the estimates are too scarce and that we do not understand yet all the effects surrounding these values.
- Ecological compensation: this is a broad topic, involving a large amount of additional issues that deserve to be addressed in a specific research.
- The debate surrounding the economic valuation of ecosystem services and the cost-benefit analysis. We approach these issues in the presentation of the limits of our work, but do not deepen this debate because several groups have already published studies on this topic, so the interest of our contribution on this debate would not be straightforward.

[4]This thesis was funded by EGIS Environnement (a subsidiary of Caisse des Dépôts et Consignations). It involved the production of a methodological guide (not attached to the thesis because it is now EGIS property). The guide provides a methodological framework and outlines the methods developed so that they can be replicated on several infrastructure projects (high-speed lines and highways).

# References

Boiteux (Rapport du groupe d'experts présidé par), 2001. Transport : choix des investissements et coûts des nuisances. Commissariat Général du Plan, Paris.

Broekx, S., Liekens, I., Peelaerts, W., De Nocker, L., Landuyt, D., Staes, J., Meire, P., Schaafsma, M., Van Reeth, W., Van den Kerckhove, O., Cerulus, T., 2013. A web application to support the quantification and valuation of ecosystem services. Environmental Impact Assessment Review 40, 65-74.

DEFRA. Case study to develop tools andmethodologies to deliver an ecosystems approach-Heysham to M6 link DEFRA research project nr0110. (Online) Available from: `http://randd.defra.gov.uk/Document.aspx?Document=NR0110_7329_FRA.pdf`2007.

Chevassus-au-Louis, B., Salles, J.M., Pujol, J.L., Bielsa, S., Martin, G., Richard, D., 2009. Approche économique de la biodiversité et des services liés aux écosystèmes : contribution à la décision publique. Documentation française, Paris.

Commisariat Général au Dévelopement Durable, 2014. Chiffres clés du transport- édition 2014, in: Service de l'observation et des statistiques, Repères, Paris. (Online) Available from: `http://www.statistiques.developpement-durable.gouv.fr`

Costanza R., 2008. Ecosystem services: Multiple classification systems are needed. Biological Conservation 141, 350-352.

Ehrlich, P.R., Ehrlich, A.H., 1981. Extinction: the causes and consequences of the disappearance of species. Random House, New York.

Géniaux, G., 1999. évaluation et régulation des impacts environnementaux. Thèse, GREQAM, Marseille.

Institut Français de l'Environnement (IFEN), 2006. L'environnement en France. MEDDE, Paris. (Online) Available from :
`http://www.statistiques.developpement-durable.gouv.fr/publications/`

SNB, (Online) Available from:
`http://www.developpement-durable.gouv.fr/-La-Strategie-nationale-pour-la-.html`

Liu, S., Costanza, R., Farber, S., Troy, A., 2010. Valuing ecosystem services. Annals of the New York Academy of Sciences 1185, 54-78.

Millenium Ecosystem Assessment, 2005. Ecosystem and well-being : A framework for assessment. Island Press Blurb, Washington DC.

Office Fédérale du Développement Territorial de Suisse, 2003. Externe kosten des verkehrs im bereich natur und landschaft Econcept, Nateco. .

Quinet, E., Baumstark, L., Bonnet, J., Croq, A., Ducos, G., Meunier, D., Rigard-Cerison, A., Roquigny, Q., 2013. Evaluation socioéconomique des investissment publics in: Française, L.d. (Ed.), Rapports et documents. Commissariat Général à la Stratégie et à la prospective, Paris.

INFRAS/IWW., 2004. External costs of transport: update study. . Schreyer, C., Maibach, M., Rothengatter, W., Doll, C., Schneider, C., Schmedding, D. The International Union of Railways (UIC) (Ed.), Karlsruhe/Zürich/Paris.

SETRA, 2010. Monétarisation des externalités environnementales. Rapport d'études, MEEDDM, Paris.

TEEB, 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. In: Kumar, P. (Ed.), Earthscan, London, Washington.

# Chapter 1

# Major challenges in mapping and managing ecosystem services in spatial planning

## Abstract

This chapter reviews the major challenges to integrate the Ecosystem Services (ES) approach into land use planning decisions, and deepens the case of terrestrial transport infrastructure implementation. This inclusion is only possible if changes in ES, involved by development policies, are explained in a spatially explicit way. Mapping ES supply, demand, and value changes in an area reveals a variety of valuable information for land use planning policies, in comparison with site-specific valuation. The representation of the ES delivered in a landscape can allow revealing environmental and socio-economic particular conflicts and synergies which in turn may help in identifying areas to be conserved in priority. ES dimensions, spatial variations and scales, are first described to highlight the need to represent ES in a spatially explicit way. Then a panorama of current available mapping techniques is drawn up, showing how these techniques try to answer different issues. From this standpoint, we show that the application of ES mapping in the transport infrastructure case bring new challenges requiring further research. We try to give an overview of the different challenges, and present the general framework surrounding the thesis.



## Résumé

Ce chapitre passe en revue les différents défis associés à l'intégration des services écosystémiques (SE) dans les décisions d'aménagement du territoire, en approfondissant le cas de l'implantation d'infrastructures de transport. L'intégration ne peut être faite que si l'estimation des SE, en termes de changements d'offre, de demande et de valeurs associées à ces changements, est faite de manière spatialement explicite. La cartographie des changements en qualité ou en quantité de SE délivrés par un territoire révèle des informations supplémentaires qui peuvent être utiles à ces choix, en comparaison aux évaluations spécifiques à un site. La représentation des SE dans l'espace permet d'identifier les aires révélant de particulières synergies ou de conflits entre les dimensions environnementales et socio-economiques du territoire, pouvant aider à mieux cibler les aires à conserver en priorité. Les dimensions spatiales des services (échelles et variations spatiales) sont d'abord décrites afin de souligner le besoin de représenter les services de manière spatialement explicite. Nous donnons ensuite un panorama des méthodes existantes permettant de cartographier les SE, en montrant comment ces techniques tentent de répondre à certains défis associés à la représentation des SE dans l'espace. À partir de cela, nous montrons que la cartographie des SE appliquée au cas des infrastructures de transport apporte de nouvelles questions nécessitant des recherches supplémentaires. Nous tentons de donner un aperçu de l'ensemble de ces questions et présentons le cadre général dans lequel s'inscrit cette thèse.

## 1.1 Introduction

Managing land use-land cover to fulfill multiple objectives of society is becoming a major challenge to policy makers. Optimal land management requires joint consideration of the value of multiple objectives. However some land-components, as Ecosystem Services (ES), are not directly traded in markets and lack readily observable signals of value. As a consequence the full variety of the consequences of land use planning choices is impossible to determine. Land-use decisions intended to maximize only observable objectives are likely to promote a decline in services delivery and consequently in human well-being. The problem is standard in environmental economics, leading to the non-market valuation techniques enabling to evaluate non-market goods and services.

Even if non-market valuation techniques have rapidly progressed, the analysis of the ES spatial dimension and distribution in the landscape has lagged and emerged only very recently (Heidkamp, 2008; Kozak et al, 2011). Estimates in economic studies are mostly site specific, a-spatial, and typically mean individual economic values are extrapolated over political jurisdictions (Kozak et al, 2011), assuming that the value do not vary within the area. A growing number of papers underline the lack of understanding on ecosystem process and functions delivering ES, demand, values derived, and how they are affected by changes in land use (Haynes-Young and Potschin, 2010; Polasky et al, 2011). The spatial economic valuation increases difficulties, because it requires extensive information, accurate data, and additional analysis. However the representation of natural capital in space would provide a finer indicator of the impact of land-use change on social welfare (Eade and Moran, 1996). Spatially sensitive valuations appear now essential to increase relevancy of the ES approach to support management planning decision.

Landscapes[1] are spatially diverse, leading to unequal distribution and unequal supply intensity of ecosystem services over an area (Willemen et al, 2010). The consideration of ES in spatial planning requires beforehand being able to assess their presence, levels and changes in a spatially explicit way, which remains by now an open challenge raising

[1]The landscape is the total spatial entity of the geological, biological and human-made environment (Hicks, 2002). Landscapes are composed of a mosaic of individual patches embedded in a matrix (Forman, 1995). The matrix comprises the wider ecosystem or the dominating land-use type e.g. agricultural, rural, or forested. Landscape patches are discrete spatial units that differ from each other due to local factors such as soil, relief, or vegetation or may also be termed "habitat". In ecology, the term habitat is a species-specific concept of the environment in which a plant or animal finds all necessary resources for survival and reproduction (Hicks, 2002).The size of a habitat is therefore entirely dependent upon the individual species' requirements.

number of issues. As Haynes-Young and Potschin, (2010) and Honrado et al, (2013) argue, the assessment should consider the dynamic relation between the ecological functions supporting ES delivery and the societal demand of each service. In order to be usable, tools and assessments must be precise reflecting at best reality, but also reproducible to connect the scientific sphere aiming at improving techniques and the real decision sphere demanding practical and applicable tools.

Here we deepen the case of the ES integration in the implementation choices related to transport infrastructure projects. The application of ES mapping in the case of transport infrastructure construction creates new opportunities to monitor and manage the services threatened by conversion. It can help raising awareness and conveys the relative importance of ES to policy makers (de Groot et al, 2012). This can also improve the efficient use of limited funds by locating areas showing a particular synergy between environmental and social dimensions ("win-win" areas) for conservation, or identify locations where ES flows can be protected at lowest cost (Crossman and Bryan, 2009; Naidoo and Ricketts, 2006). However, this consideration also leads to new challenges beyond the ones appearing in general mapping exercises. New challenges are mainly related to the conversion shape (significant length and linear) related to the infrastructure, and to projects processes defining progressively the study area and the project area. In this chapter, we try to give an overview of the issues and researches needed to map and manage ES in this particular case of development project.

The remainder of the chapter is organized as follows. Section 1.2 presents the spatial characteristics of ES. Section 1.3 presents a literature review on the current practices in spatial assessment and mapping ES supply, demand and values. Section 1.4 underlines the challenges and issues related to mapping of ES and economic value loss in the case of terrestrial transport infrastructure construction. This section presents a general approach to value spatially sensitively the loss by taking into account successively each spatial issue, from ecological to social and economic issues. The section ends by presenting the contribution of the thesis to perform this assessment. Finally section 1.5 concludes.

## 1.2 Spatial variation of ecosystem services

As any other good or service, the economic value derived from a change in quantity or quality of an ecosystem service (ES) is determined by its supply (service delivery by ecosystems) and demand (use or non-use from beneficiaries). The level of ES supply and demand are influenced by several geographical components and their extent by different scales at which they are occurring.

On the supply side, biophysical structures and ecosystem functions can vary due to ecosystem type, land use, land configuration, climate variables, hydrology, soil conditions, fauna or topography (Nelson et al, 2009; Burkhard et al, 2012). On the demand side, ES demand depends on the number of beneficiaries and their location (urban proximity), socio-economic context (income, gross domestic product per capita), preferences and social practices. Socio-economic characteristics of beneficiaries are not explicitly spatial variables per se, but differences between beneficiaries can be defined in a spatial manner (Brander et al, 2010). Issues of return to scales may also be underlined. We know from microeconomics foundations, that the incremental value granted to an abundant good or service is decreasing when quantity increases (as marginal utility declines). Conversely, a scarce demanded good may hold a high marginal value, and this value increases whereas the quantity decreases. In our setting, the abundance of the ES is met when the ecosystem is large-sized or when surrounding ecosystems supplies the same service, constituting substitutes. This dimension involves that the surface reduction of a large-sized ecosystem (or abundant in the area) providing a particular service induces a lower loss than the reduction of surface of a small-sized ecosystem (or relatively scarce in the area).

As a result, and as Boyd (2008) argues, to determine the ES values, three things really matter: location, location, location. Just like the value of a house, the value derived from ES will depend on its own quality and on its neighbourhood. Furthermore, ES occur at different spatial scales and their consideration is now recognized as an important issue to the valuation process. This fact has been well depicted in the publication of Hein et al (2006).

Ecosystems themselves vary in spatial scales as they can have a shape of small individual patches, large continuous areas or regional networks. Services delivery is generated at a range of ecological scales. As it is usually required to define the scale of a particular analysis, it has become common practice to distinguish spatially defined *ecological* scales, such as global scale, landscape scale, plot-plant or even to include the ecosystem itself as a particular scale. The most relevant ecological scale per ES has been identified, adapted from Hein et al (2006), and is presented in Table 1.1.

**Table 1.1:** Most relevant ecological scales for ecosystem services (adapted from Hein et al, 2006)- *note that some services may be relevant at more than one scale*

| Ecological scale | Dimension ($km^2$) | Services |
|---|---|---|
| Global | $>$ 1,000,000 | Global climate regulation |
| | | Carbon sequestration and storage |
| | | Regulation of albedo, temperature and rainfall patterns |
| Landscape, | 10,000-1,000,000 | Freshwater provisioning |
| Biome, | | Flood regulation |
| Watershed | | Regulation of water flows |
| | | Regulation of erosion and sedimentation |
| | | Regulation of species reproduction |
| | | Nutrient retention |
| | | Hunting recreation |
| | | Pollination |
| | | Aesthetic information and Recreation |
| Ecosystem | 1-10,000 | Food and raw material provisioning |
| or Plot-plant | $<$1 | Air quality regulation |
| | | Local climate regulation |
| | | Waste treatment |
| | | Biological control |
| | | Freshwater Fishing recreation |

As for the service delivery, demand can occur at a range of *socio-economic* scales. The assessment of a change requires identifying at which scale, and to whom the benefits of the systems' services particularly accrue. For services demanded locally, it is considered in the literature that there is a spatial limit from which the individuals do not benefit from the service anymore. When goods are local and "ordinary", some papers have proven that they are distance dependent, presenting a distance decay effect (Bateman et al, 2006; Schaafsma et al, on 2012). Distance decay highlights two important aspects: (1) it delineates the ES demand zone *i.e.*, the distance from which the ES will not be demanded anymore; (2) it indicates the spatial decay rate at which Willingness To Pay (WTP) declines when distance increases.

Distance decay is explained by different factors. The first factor is an effect of knowledge making that individuals living near a site are more likely to have knowledge about it and conversely. Besides, this is explained by the fact that the more distant a site is, the higher the costs to reach it are increased, as well as the substitutes sites' availability, decreasing net benefits (Bateman et al, 2006 ; Hanley et al, 2003). Hence, this applies particularly for direct uses made from goods and services. Indirect uses do not necessarily imply a travel, however the benefits derived from these services can be local depending on the extent of the service delivery and on the beneficiaries proximity.

The spatial extent of the demand for non-uses is supposed to be more important than those for uses because it does not require travel for beneficiaries (Bateman et al, 2006; Heidkamp et al, 2008; Schaafsma et al, 2012). This is particularly true for remarkable goods and services (existence of symbolic species, national parks), which can be considered as global (Costanza, 2008). Indeed in that case, goods are globally known and do not have or have few substitutes. They are considered as distance independent.

The surfaces on which the supply and demand meet are called the "market" area of the service (Heidkamp et al, 2008). Scales and stakeholders are often correlated, as the scale at which the ES is supplied determines which stakeholders may benefit from it. For example, when a service is supplied at the global scale, the loss or gain engendered by a project or policy changing the ES quality or quantity, will concern all the human beings because all benefited from it. For conceptualizing the relationship between the supply side and the demand side, one can imagine two overlaid maps (Brander et al, 2010): one representing the spatial extent of the potential service delivered by an ecosystem, in a given quantity and quality, according to its ecological and spatial conditions; and another one representing the spatial extent of potential beneficiaries given their preferences, the distance to the environmental good and the spatial context.

## 1.3 Spatial assessment of ecosystem services: current practices

Mapping ES research is recent but the literature has been growing substantially in the last ten years (Maes et al, 2012; Shägner et al, 2013). The increased research interest is primarily due to the public sector addressing now explicitly the use of ES maps as a crucial stage to achieve goals related to biodiversity and ES conservation (as in the case of the EU biodiversity strategy to 2020). Another reason explaining the emergence of ES mapping is that the business sector now tries to comply with future policies by developing tools and methodologies.

A recent special issue in the new *Ecosystem Services* journal provides a good snapshot on the usefulness of ES mapping and modeling ES for science and policy making (Burkhard et al, 2013: Ecosystem Services 4). The arguments supporting ES mapping are diverse in the literature and include: its usefulness to present complex and multiple information in a context in which the study of multiple ES is quite rare (Egoh et al, 2008) and its pedagogical character for communication. Further, ES mapping enables highlighting areas with particular biodiversity and ES provision interest allowing to distinguish ranges and hotspots. This can lead to an improved identification of suitable policy measures that would improve their targets and effectiveness by evaluating their benefits in relation to their costs. Mapping ES can finally be useful for green accounting, resource allocation and payments for ES (Shägner et al, 2013).

Two approaches can be distinguished in the ES modeling and mapping literature: ES modeling performed mainly by ecologist, hydrologists or geographers; and ES value mapping performed mainly by economists. The former aims at spatially representing ecosystem capacity to deliver ES by using geographical information, remote sensing data and different ecological models to generate spatially explicit maps (Egoh et al, 2008; Kareiva et al, 2011; Naidoo et al, 2008; Nelson et al, 2009; Polasky et al, 2005). The latter is typically based on non-market valuation techniques to assess and spatially differentiate the effect of land use changes on welfare (Bateman et al, 1999; Costanza et al, 1997; Eade et Moran, 1996; Kreuter et al, 2001; Troy and Wilson, 2006).

For both approaches, the precision level and representativeness of maps varies regarding data availability and quality, methodologies and models developed, or scope of the study (precisions increasing when the study scale becomes more local). Approaches are rarely but increasingly applied in a combined manner. They usually do not fully account for the associated environmental impacts and the related social welfare changes (Liekens et al, 2013).

### 1.3.1 Mapping ecosystem services supply

Four techniques can be distinguished in modeling ES supply. They differ by the level of precision and data requirement.

The first technique is a qualitative assessment based on expert opinion, professional judgment and rankings used when primary data for the study region are not available (Baral et al, 2013). Qualitative assessments mainly use participatory mapping tools, expert views converted in indicators representing professional judgments on ES conditions and temporal trends. Qualitative indicators (such as : high, moderate or low provision of ES and increasing, decreasing or stable trends) are used and transferred into Geographic Information System (GIS) to produce maps (Haines-Young et al, 2012; Scolozzi and Geneletti, 2012). The results of such analysis are criticized because of their subjectivity as they depend on the knowledge and experience of the experts for a particular landscape.

The second technique is based on ES one dimensional metrics (*e.g.* tons of carbon per hectare, tons of timber per hectare). There are many different kinds of ES and therefore different metrics are used to monitor them (for a review on proxies, see de Groot et al, 2010). Provisioning services are the easiest to put on map with representative data because they are directly quantifiable (particularly for raw materials provisioning) and, most of the time, these data are readily available in national statistics (Maes et al, 2012). However, metrics for other services are lacking, and are likely to be less reliable. Regulation and cultural services are less directly quantifiable and ecosystem capacities have to be approached by proxies as ecosystem components (environmental and spatial data, information on habitats, biodiversity, etc.) (UNEP-WCMC, 2011). Proxies are mainly used for large scale assessments.

The third technique is used when representative data is unavailable at time and places, or when services delivery depends on multiple ecosystem functions (as it is the case for ES supplied at the landscape scale). This technique relies on ecological models based on GIS tools. GIS allow manipulating, storing, analyzing large sets of geographically referenced data. One approach is the ecological production function which aims at modeling the ES output, supplied by an ecosystem given its conditions and processes (Kareiva et al, 2011; Naidoo et al, 2008; Nelson et al, 2009). Different spatial tools have been developed to help practitioners as ARIES[2] and InVEST[3] among others. InVEST is based on the use of ecological production function models (Nelson et al, 2009; Tallis and Polasky, 2011). An ecological production function specifies the potential ES outputs that are provided by an ecosystem. It uses geo-referenced data (*e.g.* land use and cover raster, digital elevation models, soil depths, potential evapotranspiration) and environmental information (*e.g.* plant available water content or nesting habitats for pollinators) to assess ES supply

[2]Artificial Intelligence for Ecosystem Services, for more information on the tool refer to Villa et al, (2007)

[3]Integrated Valuation of Ecosystem Services and Trade-offs

in biophysical terms. Once production functions are specified, it is possible to quantify the impact of a particular policy on the ES delivery (Boyd, 2008; Nelson et al, 2009; Polasky et al, 2009; Daily et al, 2011). Changes are evaluated by using future scenarios (potential future states of the natural environment) of land use changes (Swetnam et al, 2011), or compared to historical references. Nonetheless, the tool requires many fine ecological data difficult to find and an expert knowledge on ecological processes and GIS techniques (Maes et al, 2012). To sum up, a simplified representation of the approach is presented in Figure 1.1.

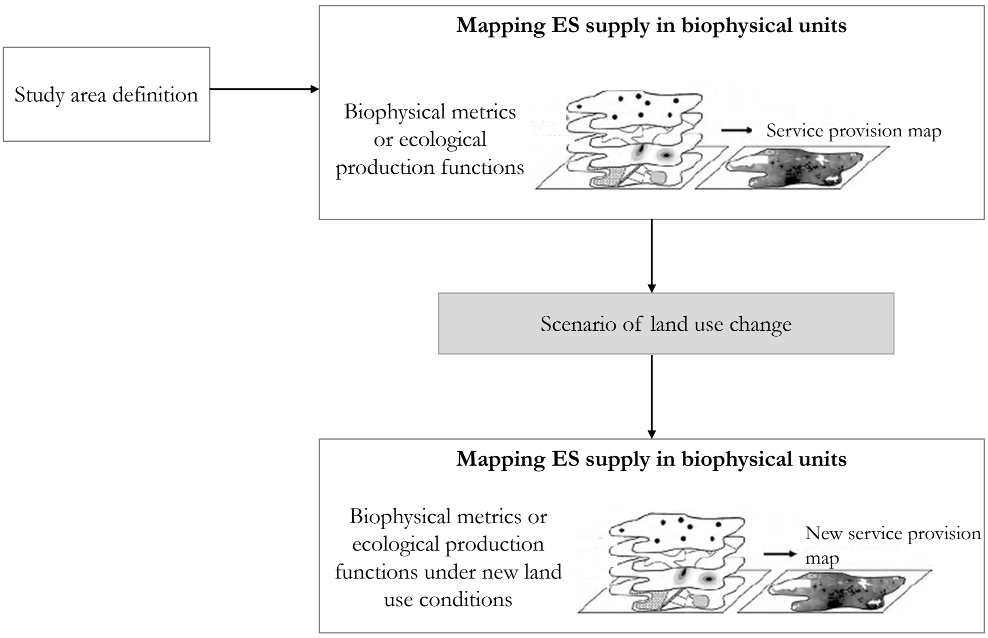



**Figure 1.1:** General approach adopted to map ES supply with ecological models

Finally, recent mapping ES supply techniques are based on biological data such as functional traits within species groups or ecosystem structures as vegetative heights, leaf dry matter contents or others detailed biological data (Lavorel et al, 2012). Such precise models principally aim at studying the relationship between biodiversity and services or trade-offs among services.

Because biophysical maps may be sufficient, when government agencies search environmental standards, economic valuation is not applied (Tallis and Polasky, 2011). Knowing how ES will change in biophysical terms is informative, but when these changes are not measured in monetary terms, as costs or benefits, it may not give full weight in decisions, because we cannot make comparisons with other costs and benefits. In this case, it can be very useful to combine biophysical assessment with economics valuation methods to estimate and report the monetary value of the change in ecosystem services quality or quantity.

### 1.3.2 Mapping ecosystem services demand and values

Non-market valuation relies on the Hicksian compensating variations[4] (or equivalent variation[5]), generally approximated by Marshallian surpluses, and measured by the beneficiaries WTP (or willingness to accept). For a given project, the compensating variation for a household is the maximum amount that it would be willing to pay to secure the change. The number is positive if and only if the project moves the (rational) household to a higher indifference surface and negative if and only if it moves the household to a lower one. A variation of satisfaction will therefore be measured by a variation in consumer surplus, and the total economic value of a good, by the total aggregated consumers' surpluses of all individuals. Sums of individuals' surpluses can be used to apply cost benefit analysis to a project or policy to assess its potential of social improvement or increase in economic efficiency.

Different primary valuation techniques have been developed to assess beneficiaries WTP for non market goods and services (see Table 1.2). Clearly, some valuation methods may be more suited to capture the values of different elements of the total economic value. For example, market prices and cost approaches are more usually used to assess provisioning services and the majority of regulation services. Revealed preference techniques might be more suitable to capture use values as recreation *e.g.* the travel cost method which uses information on the costs incurred traveling to a biodiversity-rich area to assess the recreation value of that area (Navrud and Mungatana, 1994; Shrestha et al, 2002) or landscape amenity value measured with hedonic pricing technique. Stated preference techniques would be more suited to capture non-use values *e.g.* contingent valuation method or the choice modeling can be used to assess how much people are willing to pay to protect an endangered species or habitat services (Nunes and van den Bergh, 2001).

Until recently, assessments were rarely described in a spatial manner and values were aggregated across local or large areas without the ability to determine where individuals were benefiting from the service. First applications of mapping values were conducted in

[4] This approach takes the initial level of utility as the reference point.
[5] This approach takes the final level of utility as the reference point.

**Table 1.2:** Available economic techniques for non-market valuation

| Economic techniques | Description |
|---|---|
| *Methods prices approach* | Damages are valued by using directly observed market price/or costs from actual markets as a proxy to the value |
| *Market costs approaches* | |
| Avoided damage costs | Uses the costs associated with mitigation of environmental damage as the proxy to the value |
| Replacement costs | Uses costs of replacing an environmental service as a proxy to the value |
| Opportunity costs | Explicitly considers the value that is foregone in order to protect, enhance or create a particular environmental asset |
| Production function | Focuses on the (indirect) input costs of a particular environmental service to the production of a marketed good. |
| *Revealed preferences* | |
| Travel cost method | Uses data on people's actual behaviour in real markets that are related to the environmental good. The behaviour studied is the number and distribution of trips that people make to outdoor recreation sites, as a function of, most importantly, the cost of a trip. The travel cost is the weak complement (a complementary marketed good) of the outdoor recreation value. |
| Hedonic pricing | The weak complementarity is assumed between the price of a property and the quality of the surrounding environment, the non-market value is revealed trough observations on the demand of residential properties. |
| *Stated preferences* | |
| Contingent valuation | Estimates values by constructing a hypothetical market and asking survey respondents to directly report their willingness to pay to obtain a specified good, or willingness to accept to give up a good. |
| Choice modeling | Through hypothetical market, but respondents have a series of choice tasks in which they are asked to choose their preferred policy option (including status quo). Each option is described in terms of a bundle of attributes describing the good (including a price attribute) presented at various levels according to an experimental design. The analysis of respondent choices is based on random utility maximising (RUM) theory (Hanemann, 1994). |
| *Secondary valuation technique* | |
| Benefit transfer | Uses economic information collected at a given area (study site), at a given time to make inference on environmental goods and services in another location (application site) with the same ecosystem type, at another time (Wilson and Hoehn, 2006). |

the 90's (Bateman et al, 1999; Costanza et al, 1997; Eade et Moran, 1996; Kreuter et al, 2001). Since then, the number of publications on mapping ES values has exponentially grown, with almost 60% being published after 2007 (Shägner et al, 2013). Studies quality varies according to the underlying biophysical data, land-use and land cover typology

(Global Land Cover, Corine Land Cover, or others), economic values accuracy and robustness, and on the consideration of spatial variation (Bagstad et al, 2009, Eigenbrod et al, 2010; Maes et al, 2012).

Collecting new data, and conducting primary valuations for multiple services may be very costly and time consuming. Therefore, methodological approaches for applying original valuation results in other spatial policy and decision-making contexts, usually referred to as benefit transfer, are increasingly developed and tested. Four transfer techniques are used, showing an increasing complexity and data requirement, in order to transfer benefits and estimate willingness to pay in the application site.

The simplest transfers, value transfer without adjustment, are made by applying estimates from study sites per unit area and per ecosystem type, founded in the economic literature, to the same ecosystem types in the application site (Mendoza-González et al, 2012; Kreuter et al, 2001; Troy and Wilson, 2006). Ecosystem types are located thanks to land use-land cover typologies (inland wetlands, coastal wetlands, tropical forest, temperate forests, etc.). The approach can be made following three manners: by identifying a single study which best matches with the application site and transferring a single point estimate (adjusted for inflation); by applying an average value from several studies that may better reflect the criteria by at least partially cancelling out biases in individual studies; by applying governmental approved values derived from a combination of existing empirical evidence, expert judgment, and political screening. Values transferred can also be expressed per physical unit, enabling accounting for the reality of ES supply in the application site. In some cases values transferred are expressed as a function of total economic value per ecosystem types rather than estimates of values per individual services, impeding the analysis of how provision and value of each ES will change under different conditions. The general approach of the value transfer technique (per service) is presented in the Figure 1.2.

Plummer (2009) highlighted that errors generated by this simplest mapping technique, even with individual service specific information, are likely to be high due to generalization errors. First, errors can be attributed to the extrapolation of economic values between different sites. Differences may concern social, demographic or economic information, differences in markets and substitute (Loomis and Rosenberg, 2006). In addition, errors can be done by considering the spatial constancy of biophysical measures (Bateman et al, 2006). This approach assumes indeed that the ES delivery is spatially homogeneous across ecosystem types and then, that every hectare of a given ecosystem is of equal value, regardless of flows, ecosystem spatial configuration, size or quality (Nelson et al, 2009). Spatial variation in this type of transfer depends only on ecosystem type variation.

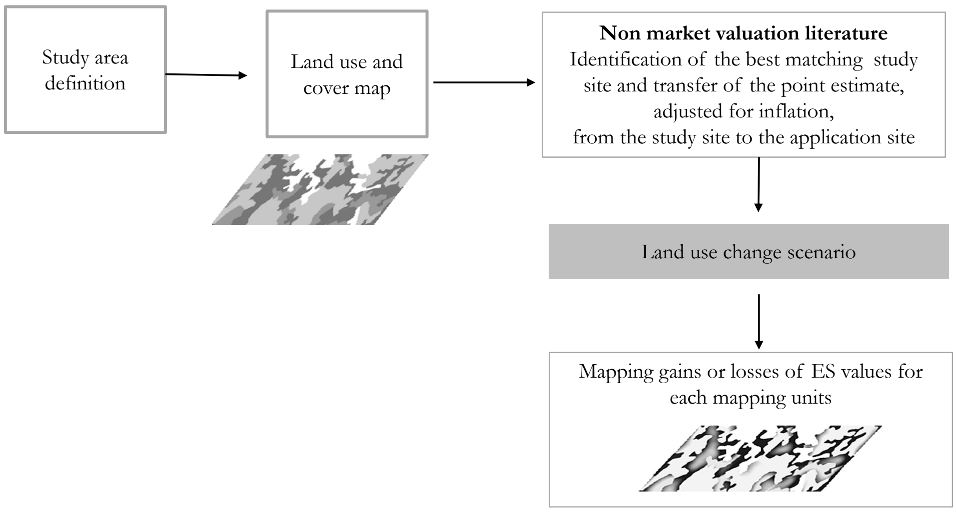



**Figure 1.2:** Mapping with value transfer approach

The application of several adjustments, reflecting differences between sites, is possible. This constitutes the second transfer technique: value transfer with adjustments. The most common adjustments are applied to correct for differences in price levels, or in income between sites. Income adjustments can for instance be done as follows (Wilson and Hoehn, 2006):

$$WTP_a = WTP_s \left(\frac{y_a}{y_s}\right)^e$$

With $y$, the income per inhabitant, $WTP$ the willingness to pay, $a$ the subscript referring to the application site and $s$ the subscript referring to the study site, and finally $e$, the income elasticity of WTP. The remaining difficulty is then related to the assessment of the level of the income elasticity of WTP. Adjustments are similar for other variables (age structure, population density, etc.). If adjustments are made across every variable determining the WTP, this is similar to the application of a value transfer function.

In the value transfer function (third transfer technique), the WTP function assessed with primary valuation techniques (travel cost, hedonic pricing, contingent valuation or choice modeling) of the study site is applied to parameter values of the application site to assess ES values. For example, a WTP function might have been estimated in which individual WTP depend on the quantity or quality of the ES provided and socioeconomic characteristics of the population originally surveyed as follows:

$$WTP_a = \beta_0 + \beta_1(ESQ_a) + \beta_2(income_a) + \beta_3(Age_a)$$

Where $ESQ_a$ represents the ES quality or quantity change in the application site $a$ that can be expressed in hectares, and the $\beta$ vector represents the regression coefficients estimated in the original study site. Bateman et al, (2011) suggest that many value function transfer exercises have failed because they have employed ad-hoc, empirically driven specification of utility functions which fit the data of study sites well but appear over-parameterized when they are applied out of sample to application sites. Due to the multiplicative role of coefficients, such parameterization can result in major transfer errors.

Last, the fourth transfer technique is the meta-analytic function that can be estimated based on multiple studies and transferred to the application site. This approach accounts for differences in results and explanatory variables in relevant studies valuing a particular ES in order to estimate a WTP function for the service. It requires collecting the available accurate studies assessing the service and codding the studies characteristics in terms of WTP estimates, non-market technique used, study site characteristics, population characteristics. From these data a regression model is estimated with WTP per unit (for a particular base year) as the dependent variable and, and at least study site characteristics, methodological attributes, and socioeconomic variables as the independent variables. The meta-regression function is then used to predict welfare estimates in the application site by inserting the levels of the independent variables that describe the policy site.

This exercise is now mainly used to "scale-up" ES values. Scaling-up is the approach of using data of existing value of local ecosystem services for an assessment of these values at a larger geographical scale: regional, national or global scale (Brander et al, 2010; Brouwer et al, 1999; Woodward and Wui, 2001). A difficulty in using this method is the multitude of original studies that may differ in at least three ways: (1) in range (changes from reference to target levels), (2) spatial and temporal scale, (3) on the number of explanatory variables that may affect the suitability of including these studies into the meta-analysis (Liekens et al, 2013). Besides, opportunities to scale-up values for land use management decisions are reduced because this technique fails in applications in multiple ES and ecosystem types, including all spatial variables. Finally, evidence from literature (Brander et al, 2010; Nelson J. et al, 2009; Smith and Pattanayak, 2002) shows that there is potentially large transfer errors and that in some cases the simple transfer of unit values may have the same performance than this less parsimonious models.

Transfer protocols have been suggested to reduce transfer errors. First, Loomis and Rosenberg (2006) prescribe to use (i) primary valuation studies based on sound scientific methods; (ii) studies having a similarity in resource conditions, site characteristics and markets, (iii) adjustments if markets are really different (*e.g.* different development levels between countries), (iv) data from meta-analysis when studies used in it are homogeneous (in protocols and goals). As an alternative for value functions estimated for transfer, Bateman et al (2011) suggest that value functions purposes should draw upon the common drivers of preferences reflected in economic theory. Such functions should avoid the problems of over-parameterization by containing only those variables which are applicable to all sites. Such drivers are: the extent of the change in provision; the costs which an individual faces for using the good (proximity of the good to the respondent's home); the availability of substitutes (again a spatial relationship) and the individual's income constraints. These variables can be assessed from secondary data (as digital maps for the site's accessibility and substitutes, census data for incomes, etc.) for both study and application sites. Finally, it can be added that using underlying biophysical models, with adapted land use-land cover data and having a scale adapted to policy needs may improve the assessment quality (Bateman et al, 2006).

### 1.3.3 Studies combining ES supply and demand

Increasing refinements of the techniques are developed in order to take into account for these protocols and for the combination of biophysical models with appropriates valuation techniques (Barbier, 2007; Bateman, 2014; Liekens et al, 2013; Naidoo et al, 2006; Nelson et al, 2009; Polasky et al, 2011). Such mapping approaches take into account the underlying mechanisms which drive ES delivery and are therefore more likely to produce realistic changes in ES supply and demand at the local and landscape scales. They enable making inference by identifying ES particular provision or the existence of distance decay, however they require significant investment in terms of data acquisition and expert knowledge.

In their study, Naidoo and Ricketts (2006) spatially evaluate opportunity costs and benefits of conservation for a landscape in the Atlantic forests of Paraguay. Services treated are bushmeat harvest, timber harvest, bioprospecting for pharmaceutical products, existence value, and carbon storage in aboveground biomass. Opportunity costs of conservation are defined as the expected agricultural value of each forested parcel of land, multiplied by the probability that a given parcel would be converted. Benefits derived from ES were calculated through biological models (as habitat associations of game species for bushmeat harvest) and economic data (essentially market prices and benefit transfer). In the same line, Polasky et al (2011), Nelson et al (2009) and Kareiva et al (2011) model ES supply (carbon sequestration and storage, water quality, agricultural production, timber production) with detailed ecological production function (InVEST tool) and apply benefit transfer or market prices to biophysical quantities across different land-uses scenarios (no agricultural expansion, no urban expansion, agricultural expansion, forestry expansion and conservation).

The United Kingdom National Ecosystem Assessment (UK NEA) values changes in ES across six different scenarios of changes for UK (that is world markets, nature at work, go with the flow, green and pleasant land, local stewardship, national security[6]). They modeled, monetized and mapped three services across UK: carbon storage, outdoor recreation and urban green-space amenity.

- Carbon storage changes were estimated for changes in above and below ground biomass (making allowance for soil type) and for fossil fuels burning to power agricultural machinery. The valuation was made through national reference values (UK official non traded carbon values) (Abson et al, 2013).

- In the case of the outdoor recreation service, a cross sectional approach has been developed, taking into account spatial determinants of the supply and demand in the WTP function. Sen et al (2014) developed a two-step model: (a) the first step aimed at predicting the number of visits estimated on the basis of interviews (48,000 interviews across 45,000 sites). They further modeled the number of visits as a function of the outset location characteristics (including population socioeconomic and demographic characteristics etc.), travel time between both sites and the destination site characteristics (including ecosystem type, surrounding potential substitutes and complements availability). In a second step (b) they determine the trip value with a meta-analytic benefit transfer.

- Urban green-space amenity was assessed by Perino et al (2014). The assessment al-

[6] For more information on scenarios definitions see Bateman et al, (2014) for a review or the UKNEA report (2011).

lows for potential marginal values to vary according to quality by identifying three types of green-space: formal urban recreation sites; informal urban green-space; and urban fringe green-space. Marginal values are then estimated as a function of a variety of determinants including green-space area, the size and income distribution of the population and location; this latter relationship follows the expected logarithmic distance decay pattern observed in other spatially sensitive valuation studies. The study aimed at scaling-up function over the UK, however a significant lack of data on size, location and quality of urban green-spaces conduced them to focus on five UK cities representing different categories of size and regional location.

Finally, recent assessments using the choice modeling technique include site characteristics to model individual heterogeneity of preferences (Abdiltrup et al, 2013; Brouwer et al, 2010; Liekens et al, 2013; Schaafsma et al, 2012). Liekens et al., (2013) use a choice experiment and a value function transfer to model and value the recreation services in Flanders. In their approach, characteristics of natural areas, beneficiaries and spatial variables such as size and distance to beneficiaries are taken into account. To do this, the choice experiment was designed in order to ask respondents to choose between different hypothetical nature development scenarios, described in terms of their ecological quality (nature type, species richness) and a set of spatial characteristics, including, size, accessibility, adjacent land use and distance to the respondent's residence. This provides information on the attributes' effect on individual's WTP. The study concerns a part of the study area, and since the transfer function is developed, it can then be scaled-up across the entire study area (Flanders).

The function developed by Liekens et al (2014) is used in Broekx et al (2013), to model and value the outdoor recreation service in Flanders. Then, Broekx et al (2013) modeled and mapped four supplementary services: carbon sequestration, nutrient retention, air quality regulation and noise mitigation by using different biological models and economic techniques (benefit transfer, hedonic pricing). From the study, they developed a web based application for Flanders (Nature Value Explorer[7]) giving a first overview of the ES loss involved by different development decisions.

Despite all these efforts, we still currently lack comprehensive studies that tie together economic valuation methods with biophysical or ecological models to estimate the monetary

[7] http://www.natuurwaardeverkenner.be

value of ES both for a broad range of ES and at a broad geographic scale.

## 1.4 Modeling and mapping ecosystem services loss for infrastructure planning

The ES mapping can be very useful in the case of transport infrastructure implementation giving a mean to widen the scope and the vision of land use planners by integrating ES considerations. An ES approach can help move project stakeholders from the confined dimension of project boundaries and regulatory check-lists, to address habitat conservation and services consideration on a broader (ecosystem or landscape) scale. Furthermore, ES approach would allow for more efficient and cost-effective ways to apply avoidance and mitigation measures which in turn would help in the capitalization on meaningful opportunities for conservation. All of this can be done through the overlaying of different landscape and ecosystem services maps, locating areas showing particular multifunctionality. In this section, we try to respond to the following question: what are the main challenges in managing and mapping ES in the case of transport infrastructure construction?
We outline the different challenges faced in such application, that brings us to present the general framework surrounding the thesis and how the different chapters approach comply the challenges laid out below.

### 1.4.1 Infrastructure construction impacts

Linear terrestrial transport infrastructures (roads, railways) can cause serious environmental impacts. In Europe, infrastructure development is identified as one of the most significant driver of habitat loss and fragmentation; other factors being intensive agriculture, industrialization and urbanization. Impacts are faced at various stages of infrastructure projects:

- During the construction: by the conversion of natural or semi-natural areas as agricultural spaces; and involving a greater fragmentation;

- By their use: disturbance from pollutants and noise emitted by vehicles; pollutants contained in the cleaning products of roads (pesticides, salting, etc.); direct collisions with the wild fauna;

- By their induced impacts, on the natural areas situated near the infrastructure, converted *a posteriori* for related developments (subsidiary development such as housing, industry, etc.).

The environmental impacts are generally classified as direct impacts and indirect impacts. The direct impacts lead to the destruction of natural or semi-natural ecosystems crossed by the infrastructure (habitat loss), by soil tillage, sealing and conversion into metaled/sealed surfaces. The infrastructure construction affects the physical environment at the edges of the infrastructure axis due to the need to clear, level, fill, and cut natural material. Construction work changes soil patterns, landscape relief, surface, groundwater flows, and micro-climate, and thus alters land cover, vegetation, ecological functions and habitat composition. Wetlands and riparian habitats are especially sensitive to changes in hydrology *e.g.* those caused by embankments (Findlay and Bourdages, 2000) and cuttings which may drain aquifers. This in turn increases the risk of soil erosion and extensive earth-slides that have the potential to pollute watercourses with sediments (*e.g.* Forman et al, 1997; Trombulak and Frissell, 2000). The surface water canalization into ditches can also significantly change water run-off and debris flows, and thereby modify disturbance regimes in riparian networks (Jones et al, 2000). The clearance of a road corridor changes local climate conditions by increasing light intensity, reducing air humidity, and creating a greater daily variation in air temperature. These changes are naturally stronger when the road crosses through forested habitats. All these micro-environmental changes are known to affect plant growth and performance (Tardieu F. et al, 2014). Chemical disturbance is also observed by accumulation of chemical pollutants around the infrastructure axis such as road dust, salt, heavy metals, fertilizer nutrients, and toxins, which contribute towards the disturbance effect caused by transportation infrastructure. The direct impacts also include the temporary disturbance of the surrounding ecosystems during the roadworks as the establishments of deposit zones for roadwork's materials and zones of secondary structures in the works (as accesses to the construction site, zone of extraction, etc.).

The area covered by roads and railways is, however, not a reliable measure of the loss of natural habitat and ecological functions. It is now well recognized, and proven by empirical studies, that the disturbance influence on surrounding wildlife, vegetation, hydrology, and landscape spreads much wider than the area that is physically occupied. This contributes far more to the overall loss and degradation of habitat than the road/railway body itself (for a review see Trocmé et al, 2002). These extended effects are generally called indirect impacts provoked by the interruption or disruption ecological flows at different scales and other disturbances in terms of noise and visual nuisance, which act to reduce the suitability of adjacent areas for wildlife (Hicks et al, 2002). However, the major driver of indirect impacts is habitat fragmentation dividing natural habitats into units of smaller size known as patches. Beyond the reduction of the size of crossed habitats, it affects the structure and functioning of the landscape as a whole by acting as a barrier to the movement and migration of animal and plant species (Quintana et al, 2010). The barrier effects is also acting through other physical, visual, thermal or chemical effects, entailing an imbalance in the meta-populations organization that endangers species survival (Vanpeene-Bruhier and Dalban-Canassy, 2006). This may result in the isolation and extinction of vulnerable species. Further, where infrastructure dissects a foraging, commuting, dispersal or migration route, animals will have to cross the barrier and encounter a higher risk of mortality from traffic impact.

Hence, at the beginning of the process, the loss of habitat is the driving force reducing biodiversity in the landscape. Towards the end of the process, isolation effects become stronger (Hicks et al, 2002).

### 1.4.2 Challenges in mapping ES loss in the case of transport infrastructure implementation

As discussed before, the incorporation of the spatial context in valuation studies brings many difficulties which can be resumed through the ES cascade framework (Figure 1.3). This general framework proposed by Haines-Young and Potschin (2010) describes the general challenges in assessing ES changes by linking ecological and socio-economic dimensions. Ecosystems provide the necessary structure and processes that underpin ecosystem functions. ES are derived from ecosystem functions and represent the realized flow of services in relation to the benefits and values of people (Maes et al, 2012).We adapted the framework for application to our case.

The spatial character of the valuation requires additional analysis for each steps and this is also true in the infrastructure construction case. General challenges are in an overlapping way: the definition of the biophysical structures, process flows in initial conditions to assess the change after the policy implementation in a spatially explicit way; the definition of ecological/biophysical models to assess changes in ecosystem functions and in services provision in quantity or quality, the identification of potential beneficiaries and their demand according to the spatial context and socio-economic characteristics (travel

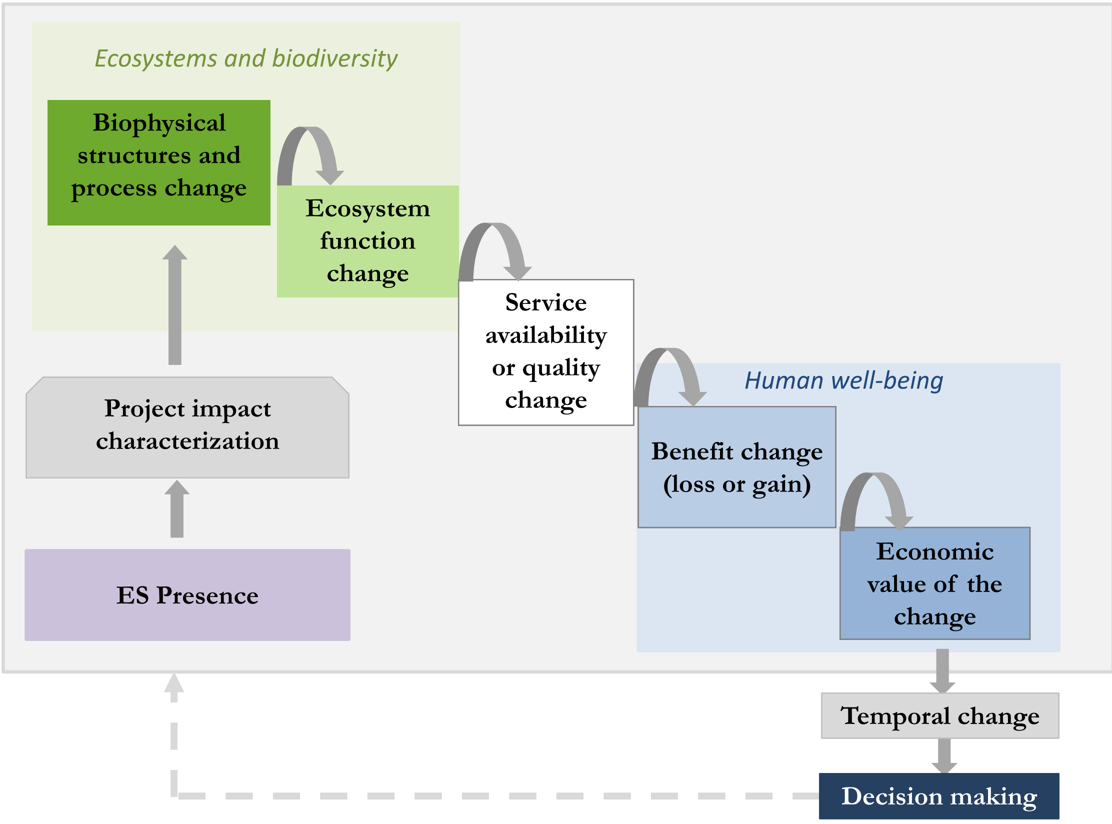



**Figure 1.3:** The ecosystem services cascade framework (adapted from Haines-Young and Potschin, 2010)

distance, substitutes); the determination of the accurate value to assess the service in accordance with the ecosystem spatial context. The process therefore requires a substantial amount of data at every step.

The application of this frame to the case of infrastructure construction leads to specific issues at every steps of the cascade framework. Issues are mainly associated to the project process and to the transport infrastructure shape that are identified below.

***Implementation options are defined progressively:***

The precision of the land use change scenario depends on when the assessment is achieved during the project process. The object of the impact analysis moves from 10-15 km wide passage corridors (during preliminary studies), to 1-5 km spindles, and finally to reference routes throughout the selected spindle of hundred meters wide. Hence, the spatial scale of geographical information systems outputs varies from 1:50 000 (1 cm for 500m) during the first project stages down to 1:5 000 (1cm for 50m) for the last stages. The infrastructure land-take (direct impact extent) is also precised during the process, passing from a hundred meters around the axis to the precise land take including carries and land arrangements. Finally, measures to reduce environmental impacts are defined and located late in the process, enabling no proper analysis of the residual impact if the assessment is made in the preliminary stages of the project. This has repercussions on the definition of biophysical structures and process changes and then on the overall assessment of the loss.

***Infrastructures cross varied multifunctional landscapes***

Transport infrastructures have significant length, especially when infrastructures are highways or high speed rails connecting large cities or countries. The significant length involves that it impacts different multifunctional landscapes, and different ecosystems providing more than one service. The mapping exercise is then even more complicated. Assessing the loss for the entire implementation option require multiple ecological data (on climate, hydrology, topography, land use and cover), socio-economic data, population repartition data in order to produce layers that effectively identify conflicts or synergies across the landscape.

***Defining the spatial extent of impacts on the flow of ecosystem services***

Terrestrial transport infrastructures construction causes an irreversible conversion of natural and semi-natural ecosystems in a linear and continuous form, causing direct and indirect impacts. We can presume multiple kinds of effects on ES having different spatial extent. The incidences on ES flows can be classified in the same order than general impacts that is in terms of *direct* ES loss involved by the conversion and additional *indirect* ES loss to impacts on landscape connectivity.

Direct losses extent will differ according to the infrastructure type and has to be defined according to the project stage in which the assessment is made (with buffer zones in the first stages, generally larger for highways than for high-speed rails, and precise project land-take in the last stages).

Landscape composition (the spatial cover of land use types in a given area) and configuration (the pattern of different elements in a landscape) can both affect the provision of ES (Mitchell et al, 2013; Ng et al, 2013). Indirect effects associated with modifications to landscape connectivity, *i.e.* the flow and movement of materials and organisms across a landscape may impact on a range of different ES. Then, the destruction of an ecosystem in an impact zone may cause a decline in ES provision by a landscape element exterior to the zone. Hence, ES loss will be function of impacts on the ecological interdependencies among the different elements of a landscape or territory. Moreover, some species and ecosystems may need a minimum surface to complete their ecological functions (Muradian, 2001; Groffman et al, 2006). Hence, when part of a habitat or ecological network is lost due to development, species responses and ecosystem function may show a non-linear response to land conversion due to threshold behaviour. Finally, when the ES supply depends on the quality of the ecosystem on a particular point, the loss can be related to whether the infrastructure is more or less distant to the point of interest (*e.g.*, recreational forest impacted by visual or noise disturbance). Hence in this case, the perturbation of a particular unique landscape element may have a large effect relative to the disturbance surface area (Van der Zee, 1990; Eade and Moran, 1996).

Disregarding all these effects is likely to conduce to an underestimation of the ES loss. The determination of which kind of flows are impacted and to what extent they are impacted is then critical in the assessment.

***Defining the degree of ecosystem services loss***

Changes in ecosystems flows can have different degrees and temporal trends according to the way they are impacted. First, we can presume a total loss for all directly impacted ES flows, over the infrastructure lifespan, as suggested by the COPI[8] project (Braat and Ten Brink, 2008). The soil below the infrastructure will be converted for a long period, and functions underlying all services will be threatened. Indeed, the COPI project relates some management systems going from natural and light use to degraded and converted use, to the magnitude of impacts on biodiversity and ES (see Figure 1.4). For provisioning services, and particularly agricultural and forestry products, it is assumed that a minimum disturbance is needed to harvest products. Then, the production grows with the intensity of the use thanks to human-made inputs (fertilizers, water, labor, pesticides). The most extreme types of land use are built up areas and areas covered for infrastructure purposes, where production of natural ecosystem goods approaches zero. For regulation services, it is supposed that the optimal production is performed under the most natural states of

[8] Cost of Policy Inaction

ecosystems, and decreasing with the intensification of the use until reaching zero for converted areas. Lastly, for recreation services, the accessibility of the recreational sites plays a crucial role in the potential benefit derived, then a minimum disturbance is considered as required. However, the conversion involves also a total loss.

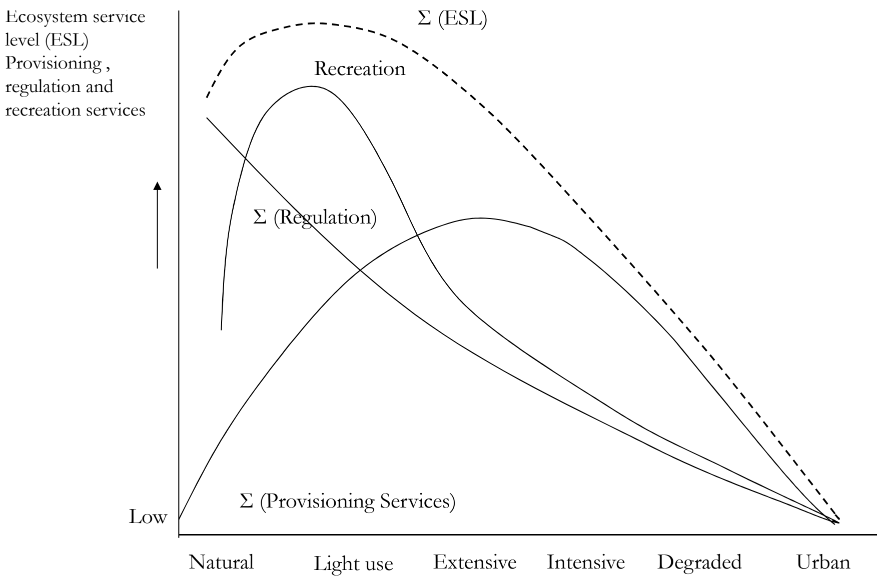



**Figure 1.4:** Generalized functional relationships between the levels of ecosystem services provision and the land management (adapted from Braat and ten Brink, (2008) and de Groot et al, (2010))

For the indirect losses suffered around the infrastructure axis, we can presume that the ecosystem can recover its service flows during the infrastructure lifespan thanks to its resilience capacities. The characterization of the impact should then take into account for this possibility. Beyond the ecosystem resilience, impacts on natural areas are reduced through environmental measures (*e.g.* providing links between habitats severed by the infrastructure as wildlife crossing structures or fauna passages), hence some of ES impacted will not be totally lost, over all the lifespan period, in the vicinity of the infrastructure.

***Dealing with the temporal trend of the ES loss***

When the ES flows are considered as lost, questions of how to discount values over time arise. On the one hand, the costs and benefits associated with these long-term consequences tend to see their weight in decisions minored when time discounting is applied. On the other hand, there is an uncertainty about the future consequences of the losses and on the behaviour and expectations of future generations for some natural assets. This raises the problem of choosing a discount rate that can be applied to environmental goods and services.

***The analysis of ES loss should be exhaustive***

Conversely to other spatial assessments focusing on one particular ecosystem delivering multiple services, or in a particular service delivered by multiple ecosystems, the assessment needs to be, as far as possible, exhaustive in order to assess multiple services of multiple ecosystems. This must be done because we have to compare multiple implementation options for the infrastructure. This implies achieving the ES cascade framework for a range of services and a range of ecosystems. Moreover, if the assessment is made over one implementation option, it has to be made over the other ones in order to be able to compare them. Finally, if the service is assessed over one ecosystem type, it has to be performed over the other types of ecosystems that deliver the service in order to not discriminate an ecosystem with respect to another because of insufficient data.

***The analysis of ES loss should be discriminant***

Even if the spatial assessment needs to be as far as possible exhaustive, implementation options are likely to be geographically close, thus the analysis needs to be sufficiently precise in each step of the cascade framework to enable the discrimination between options. This can be achieved by using precise Land-Use/Land-Cover (LULC) typologies and by defining spatial conditions of service's (potential) presence according to ecosystems types and their locations when ecological models are not used[9]. The ability to make linkage between services and LULC typologies is a real issue. To our knowledge two studies bound one of the most used typology *i.e.* Corine Land Cover typology (2006) to ecosystem services potential presence (Burkhard et al, 2009; CGDD, 2010). However the more precise the typology is, the less likely it is to find accurate data for it. This can also be made through an accurate ES supply mapping technique. And finally, this can be done by taking into account beneficiaries and their spatial context. If no data are available, some spatial assumptions can be made on the likelihood of service demand (*e.g.* air quality regulation is performed by forests situated in urban or peri-urban areas and used by population in these areas).

[9]Ecological models calculates the presence given the data. When no models are used to determine the presence, spatial conditions of presence can be specified: *e.g.* regulation of water flows is delivered by a wetland only if it is related to a river system.

***Balancing scientific reliability, reproducibility and user friendliness***

Accounting for ecosystem services in a spatial manner is complex, costly and time consuming and as we have seen in particular when it comes to the case of infrastructure projects. Balancing scientific reliability (list of services covered, uncertainty analysis, precise methodologies for mapping) against user-friendliness and replicability of methods remains a big challenge as this reproducibility is also required.

### 1.4.3 Contribution of the thesis to face some of these challenges

When dealing with ES analysis in the case of transport infrastructure projects, a number of issues remain related to the general mapping exercises or to the particular challenges associated to this specific case. It requires a multi-scale approach, a broader assessment and defining the particular treatments to apply for infrastructure disturbance on ES flows.

Multiple steps can be defined from the cascade framework to perform the assessment by taking into account spatial variables. The first step aims at identifying ES potential presence given the LULC typology used and spatial assumptions specified for the presence. The second step seeks to build a scenario of land use change depending on the available data on implementation option, and on the form of loss according to the service impacted correlated to the way it is delivered by ecosystems. The third step consists in analysing the consequences of the changes in the potential provision of services on the basis of supply models or proxies and given the importance of the impact considered. The assessment approach may be determined by the ES supply scale *i.e.* when the most relevant ecological scale for the ES provision is the landscape, ecological production functions may be more relevant reflecting better the interaction between ecosystems; when conversely services are supplied at the ecosystem scale, proxies may be sufficient. The influence of the beneficiaries' location or the spatial context's influence on the value may also be taken into account to assess the loss. At that point the long term assessment is possible through making explicit assumptions on the future trend of ES supply and demand. Finally, the information may be submitted to projects practitioners and decision makers through adapted instruments to current projects assessment tools.

The following chapters presented in this thesis approach face these challenges. In each essay we apply the framework in a real case study of a high speed rail project in Western France. Chapter 2 applies the cascade framework in the case of a single service, which is a “simple” service in terms of impact characterization and of ES modeling and mapping, that is the global climate regulation service. This was made in order to test the feasibility of such an assessment. The presence of the service can be deduced directly from the land use-land cover data. Then the loss can be presumed to be direct (contained in the infrastructure width), and the variation can be assessed trough the carbon stocks proxy (in ton of carbon per hectare). The service being global (for the supply and demand) we can presume no influence of beneficiaries' location or spatial context. In this case the use of constant value applied to physical quantities (varying in space) can be considered as acceptable.

Other types of services require more detailed GIS functions to successively treat spatial effects. This is the main topic of Chapter 3 applying the cascade framework for a range of services (flood protection, erosion control, pollination, etc.), and identifying the different impacts of the infrastructure on ES flows in terms of direct and indirect loss. The assessment is made at the stage of comparison of implementation options. Assumptions for ES presence are explicitly defined, a typology of the project impacts and impact zone on ES flows is drawn, the use of multiple ES supply assessment is performed as well as the economic valuation in order to assess and map ES loss for each implementation options.

Last, Chapter 4 concentrates on the last part of the cascade framework, that is on the relevance of the ES analysis for transport planning. It deals with the integration of such analysis in the current legal framework surrounding transport project evaluation, into the current evaluation tools. To do this, a thinking on how values can be expressed to inform decision-makers at each stage of the project is conducted (this may be done through an indicator, trough annual economic values, or through long-term losses). To express the loss in a long-term horizon, the definition of temporal trends for each ES values needs beforehand to be done. The test of ES accounting to assess the cost-efficiency of mitigating measures is also made. We show that this analysis can provide guidance at different stages of transport project: from the preliminary studies to the study of the final implementation option. As a result, this may help project stakeholders to better consider the effects of the infrastructure implementation, and to better target environmental measures.

## 1.5 Conclusion

ES mapping is becoming an important and powerful tool to support day-to-day land management decisions. It should allow for the location of accurate win-win areas to be conserved in priority. This type of analysis can be particularly important in the assessment of ordinary natural areas, not protected by any conservation status. However, ES mapping stays in its infancy and requires many research advances to improve techniques and knowledge on the processes involved. Ecosystem services valuation stays criticized for reasons of consistency, and the ES value mapping is no exception for this general rule.

For an overview on the ES mapping literature, we can rely on the Shağner et al (2013) review. They summarized the recent literature on mapping ES values published until the year 2012, and identified 69 papers on Scopus, Science Direct and Google Scholar. They found that the recreation has been the service most often mapped (50 studies), followed carbon storage and sequestration service (40 studies). Most publications mapped one individual service (28%), and more than 50% mapped three or less services. The average number of mapped ecosystem services per study is 7 ES[10]. The majority of ES supply mapping was based on proxies (52%), and 84% of the studies used value transfer at least for one service (with 80% using unit value transfer).

Accounting for the determinants of both ES supply and its values for a range of services supplied by a range of ecosystems requires a deeper integration of the disciplines involved (Bockstael et al, 2000). Few studies combine the strengths of multiple perspectives. Studies that are dominated by an ecological perspective tend to use sophisticated ES models, but fails in an accurate economic valuation. Conversely, studies that are dominated by an economic perspective may focus on the valuation process, but tend to rely on proxies or on a value implicitly holding a signal of ES supply for ES quantification. For a decision making perspective hence, despite its undeniable usefulness, maps should be used with caution and be interpreted after having examined all the whys and wherefores besides resulted maps (Hauck et al, 2013). The selection of the most appropriate mapping approach must also be questioned.

The barriers related to the development of highly accurate ES value maps are manifold.

[10]The study of Costanza et al (1997) mapped 17 ES at the global scale (with indicators), and their approach has been replicated several times.

They concern essentially:

- a lack of data in quantity or in quality (spatial resolution): ES mapping requires a significant quantity of data in LULC, population, environmental data (climatic, hydrological, topography, others), built capital, ecosystem status (private or public), agricultural production, present state (*e.g.* present pollution concentration);

- a lack of knowledge in biodiversity and ecosystem processes: interaction between biodiversity and ES *i.e.* how an incremental change impacts the ES supply, the interaction between land use and ES provision and tradeoffs between ES;

- a lack of understanding in ecosystem impact resistance: the determination of the threshold from which ecosystems shift into a critical state can be crucial for ES valuation and policy recommendation;

- a lack of knowledge on individuals' spatial preferences, and a lack of possibility to integrate these variations under some valuation techniques (unit value transfer, cost and market price approaches);

- and finally a lack of comprehension in the effect of combining multiple method to assess benefits derived from ES, and on their spatial aggregation.

The application of ES loss mapping to the transport infrastructure project construction case is carrying new challenges. By converting and fragmenting landscape, linear infrastructures modify the ecosystem ecological functions and processes, ES supply and benefits derived from those services. Spatial factors take a critical importance to assess accurately ES loss induced by the construction. In this chapter, we highlight the methodological issues related to a spatially explicit mapping of ES loss. By now, infrastructures implementation case has been only taken as examples of cases studies for the use of ES mapping. To our knowledge only one peer-reviewed paper assessed the loss involved by infrastructure construction, Broekx et al, (2013) as a case study of their web-based tool. However, issues of ES scale or impact zones are not addressed. Hence, there is still much to do in this regard.

## References

Abildtrup, J., Garcia, S., Olsen, S.B., Stenger, A., 2013. Spatial preference heterogeneity in forest recreation. Ecological Economics 92, 67-77.

Abson, D., Termansen, M., Pascual, U., Aslam, U., Fezzi, C., Bateman, I., 2014. Valuing Climate Change Effects Upon UK Agricultural GHG Emissions: Spatial Analysis of a Regulating Ecosystem Service. Environmental and Resource Economics 57, 215-231.

Bagstad, K.J., 2009. Ecological economic applications for urban and regional sustainability, Natural Resources. University of Vermont, Burlington.

Baral, H., Keenan, R.J., Stork, N.E., Kasel, S., 2013. Measuring and managing ecosystem goods and services in changing landscapes: a south-east Australian perspective. Journal of Environmental Planning and Management, 1-23.

Bateman, I., Brouwer, R., Ferrini, S., Schaafsma, M., Barton, D., Dubgaard, A., Hasler, B., Hime, S., Liekens, I., Navrud, S., De Nocker, L., Sceponaviciute, R., Semeniene, D., 2011. Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe. Environmental and Resource Economics 50, 365-387.

Bateman, I., Harwood, A., Abson, D., Andrews, B., Crowe, A., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., Termansen, M., 2014. Economic Analysis for the UK National Ecosystem Assessment: Synthesis and Scenario Valuation of Changes in Ecosystem Services. Environmental and Resource Economics 57, 273-297.

Bateman, I.J., Day, B.H., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values: Welfare measures, distance decay and total WTP. Ecological Economics 60, 450-460.

Bateman, I.J., Ennew, C., Lovett, A.A., Rayner, A.J., 1999. Modelling and mapping agricultural output values using farm specific details and environmental databases. Journal of Agricultural Economics 50, 488-511.

Bockstael, N., E.„ Freeman, A.M., Kopp, R.J., Portney, P.R., Smith, V.K., 2000. On measuring economic values for nature. Environmental Science & Technology 34, 1384-1389.

Boyd, J., 2008. Location, location, location: the geography of ecosystem services. Resources for the Future, 11-15.

Braat, L., ten Brink, P., 2008. The Cost of Policy Inaction: The case of not meeting the 2010 biodiversity target. in: European Commission, D.E. (Ed.), Brussels/Wageningen.

Brander, L.M., Ghermandi, A., Kuik, O., Markandya, A., Nunes, P.A.L.D., Schaafsma, M., Wagtendonk, A., 2010. Scaling Up Ecosystem Services Values: Methodology, Applicability and a Case Study. FEEM Working Paper No. 41.

Broekx, S., Liekens, I., Peelaerts, W., De Nocker, L., Landuyt, D., Staes, J., Meire, P., Schaafsma, M., Van Reeth, W., Van den Kerckhove, O., Cerulus, T., 2013. A web application to support the quantification and valuation of ecosystem services. Environmental Impact Assessment Review 40, 65-74.

Brouwer, R., Langford, I.H., Bateman, I.J., Turner, R.K., 1999. A meta-analysis of wetland contingent valuation studies. Regional Environmental Change 1, 47-57.

Brouwer, R., Martin-Ortega, J., Berbel, J., 2010. Spatial Preference Heterogeneity: A Choice Experiment. Land Economics 86, 552-568.

Burkhard, B., Crossman, N., Nedkov, S., Petz, K., Alkemade, R., 2013. Mapping and modelling ecosystem services for science, policy and practice. Ecosystem Services 4, 1-3.

Burkhard, B., Kroll, F., Müller, F., Windhorst, W., 2009. Landscapes´ Capacities to Provide Ecosystem Services- a Concept for Land-Cover Based Assessments. Landscape Online 15, 1-22.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators 21, 17-29.

Commissariat Général au Développement Durable (CGDD), 2010. Projet de caractérisation des fonctions écologiques des milieux en France. Etudes et Documents 20-2010. Paris.

Costanza, R., d´ Arge, R., Groot, R.D., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., Belt, M.v.d., 1997. The value of the worldś ecosystem services and natural capital. Nature 387, 253-260.

Costanza R., 2008. Ecosystem services: Multiple classification systems are needed. Biological Conservation 141, 350-352.

Crossman, N.D., Bryan, B.A., 2009. Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. Ecological Economics 68, 654-668.

Daily, G.C., Kareiva, P., Polasky, S., Ricketts, T., Tallis, H., 2011. Mainstreaming natural capital into decisions, in: Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E. (Eds.), Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, Oxford, UK.

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L.C., ten Brink, P., van Beukering, P., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services 1, 50-61.

Eade, J.D.O., Moran, D., 1996. Spatial Economic Valuation: Benefits Transfer using Geographical Information Systems. Journal of Environmental Management 48, 97-110.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. Agriculture, Ecosystems & Environment 127, 135-140.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. Journal of Applied Ecology 47, 377-385.

Findlay, C.S., Bourdages, J., 2000. Response time of wetland biodiversity to road construction on adjacent lands. Conservation Biology 14. 86-94.

Forman, R.T.T., 1995. Land mosaics: The ecology of landscapes and regions. Cambridge University Press, Cambridge.

Forman, R.T.T., Friedman, D.S., Fitzhenry, D., Martin, J.D., Chen, A.S., Alexander, L.E., 1997. Ecological effects of roads: Towards three summary indices and an overview for North America, in: Canters, K., Piepers, A., Hendriks-Heersma, A. (Eds.), Proceedings of the international conference on Habitat fragmentation, infrastructure and the role of ecological engineering. The Netherlands: Ministry of Transport, Public Works and Water Management, Road and Hydraulic Engineering division., Delft, pp. 40-54.

Geneletti, D., 2011. Reasons and options for integrating ecosystem services in strategic environmental assessment of spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management 7, 143-149.

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., LeRoy Poff, N., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1-13.

Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being, in: Raffaelli, D., C., F. (Eds.), Ecosystem Ecology: a new synthesis. BES Ecological Reviews Series. Cambridge University Press, Cambridge pp. 110-139.

Hanemann, W.M., 1994. Valuing the Environment through Contingent Valuation. Journal of Economic Perspectives 4, 19-44.

Hanley, N., Schlapfer F., J., S., 2003. Aggregating benefits of environmental improvements: distance-decay functions for use and non-use values. Journal of Environmental Management 68, 297-304.

Hauck, J., Görg, C., Varjopuro, R., Ratamäki, O., Maes, J., Wittmer, H., Jax, K., 2013. "Maps have an air of authority": Potential benefits and challenges of ecosystem service maps at different levels of decision making. Ecosystem Services 4, 25-32.

Heidkamp, C.P., 2008. A theoretical framework for a spatially conscious economic analysis of environmental issues. Geoforum 39, 62-75.

Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecological Economics 57, 209-228.

Hicks, C., Peymen, J., 2002. Habitat Fragmentation due to Existing Transportation Infrastructure, in: Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J. (Eds.), COST 341 - Habitat Fragmentation due to transportation infrastructure: The European Review. Office for Official Publications of the European Communities, Luxembourg, pp. 73-113.

Honrado, J.P., Vieira, C., Soares, C., Monteiro, M.B., Marcos, B., Pereira, H.M., Partidário, M.R., 2013. Can we infer about ecosystem services from EIA and SEA practice? A framework for analysis and examples from Portugal. Environmental Impact Assessment Review 40, 14-24.

Jones, J.A., Swanson, F.J., Wemple, B.C., Snyder, K.U., 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14, 76-85.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E., 2011. Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, Oxford, UK.

Kozak, J., Lant, C., Shaikh, S., Wang, G., 2011. The geography of ecosystem service value: The case of the Des Plaines and Cache River wetlands, Illinois. Applied Geography 31, 303-311.

Kreuter, U.P., Harris, H.G., Matlock, M.D., Lacey, R.E., 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecological Economics 39, 333-346.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011. Using plant functional traits to understand the landscape distribution of multiple ecosystem services. Journal of Ecology 99, 135-147.

Liekens, I., Schaafsma, M., De Nocker, L., Broekx, S., Staes, J., Aertsens, J., Brouwer, R., 2013. Developing a value function for nature development and land use policy in Flanders, Belgium. Land Use Policy 30, 549-559.

Loomis, J.B., Rosenberger, R.S., 2006. Reducing barriers in future benefit transfers: Needed improvements in primary study design and reporting. Ecological Economics 60, 343-350.

Maes, J., Egoh, B., Willemen, L., Liquete, C., Vihervaara, P., Schägner, J.P., Grizzetti, B., Drakou, E.G., Notte, A.L., Zulian, G., Bouraoui, F., Luisa Paracchini, M., Braat, L., Bidoglio, G., 2012. Mapping ecosystem services for policy support and decision making in the European Union. Ecosystem Services 1, 31-39.

Mancebo Quintana, S., Martín Ramos, B., Casermeiro Marínez, M.A., Otero Pastor, I., 2010. A model for assessing habitat fragmentation caused by new infrastructures in extensive territories - Evaluation of the impact of the Spanish strategic infrastructure and transport plan. Journal of Environmental Management 91, 1087-1096.

Mendoza-González, G., Martínez, M.L., Lithgow, D., Pérez-Maqueo, O., Simonin, P., 2012. Land use change and its effects on the value of ecosystem services along the coast of the Gulf of Mexico. Ecological Economics 82, 23-32.

Millenium Ecosystem Assessment, 2005. Ecosystem and well being : A framework for assessment. Island Press Blurb, Washington DC.

Mitchell, M.E.G., Bennett, E.M., Gonzalez, A., 2013. Linking landscape connectivity and ecosystem service provision: current knowledge and research gaps. Ecosystems 16, 894-908.

Muradian, R., 2001. Ecological thresholds: a survey. Ecological Economics 38, 7-24.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Ecosystem Services Special Feature: Global mapping of ecosystem services and conservation priorities. PNAS 105, 9495-9500.

Naidoo, R., Ricketts, T.H., 2006. Mapping the Economic Costs and Benefits of Conservation. PLoS Biol 4 (11), e360.

Navrud, S., Mungatana, E.D., 1994. Environmental Valuation in Developing Countries: The Recreational Value of Wildlife Viewing. Ecological Economics 11, 135 - 151.

Nelson, J., Kennedy, P., 2009. The Use (and Abuse) of Meta-Analysis in Environmental and Natural Resource Economics: An Assessment. Environmental and Resource Economics 42, 345-377.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7, 4-11.

Ng, C.N., VXie, Y.J., Yu, X.J., 2013. Integrating landscape connectivity into the evaluation of ecosystem services for biodiversity conservation and its implications for landscape planning. Applied Geography 42, 1-12.

Nunes, P., van den Bergh, J., 2001. Economic Valuation of Biodiversity: Sense or Nonsense? Ecological Economics 39, 203-222.

Perino, G., Andrews, B., Kontoleon, A., Bateman, I., 2014. The Value of Urban Green Space in Britain: A Methodological Framework for Spatially Referenced Benefit Transfer. Environmental and Resource Economics 57, 251-272.

Plummer, M.L., 2009. Assessing benefit transfer for the valuation of ecosystem services. Frontiers in Ecology and the Environment 7, 38-45.

Polasky, S., Nelson, E., Pennington, D., Johnson, K., 2011. The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. Environmental and Resource Economics 48, 219-242.

Polasky, S., Segerson, K., 2009. Integrating Ecology and Economics in the Study of Ecosystem Services: Some Lessons Learned. Annual Review of Resource Economics 1, 409-434.

Schaafsma, M., Brouwer, R., Rose, J., 2012. Directional heterogeneity in WTP models for environmental valuation. Ecological Economics 79, 21-31.

Schägner, J.P., Brander, L., Maes, J., Hartje, V., 2013. Mapping ecosystem services' values: Current practice and future prospects. Ecosystem Services 4, 33-46.

Scolozzi, R., Geneletti, D., 2012. Assessing Habitat Connectivity for Land-use Planning: a Method Integrating Landscape Graphs and Delphi Survey. Journal of Environmental Planning and Management 55, 813-830.

Sen, A., Harwood, A., Bateman, I., Munday, P., Crowe, A., Brander, L., Raychaudhuri, J., Lovett, A., Foden, J., Provins, A., 2014. Economic Assessment of the Recreational Value of Ecosystems: Methodological Development and National and Local Application. Environmental and Resource Economics 57, 233-249.

Schaafsma, M., Brouwer, R., Rose, J., 2012. Directional heterogeneity in WTP models for environmental valuation. Ecological Economics 79, 21-31.

Shrestha, R.K., Seidl, A.F., Moraes, A.S., 2002. Value of Recreational Fishing in the Brazilian Pantanal: A Travel Cost Analysis Using Count Data Models. Ecological Economics 42, 289-299.

Smith, V.K., Pattanayak, S.K., 2002. Is meta-analysis a Noah's Ark for non-market valuation. Environmental & Resource Economics 22, 271-296.

Tallis, H., Polasky, S., 2011. Assessing multiple ecosystem services: an integrated tool for the real world, in: Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E. (Eds.), Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, Oxford.

Tardieu, F., Parent, B., Caldeira, C.F., Welcker, C. 2014. Genetic and physiological controls of growth under water deficit. Plant Physiology 164, 1628-1635.

Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J., 2002. COST 341 -Habitat Fragmentation due to transportation infrastructure: The European Review. Office for Official Publications of the European Communities, Luxembourg.

UK National Ecosystem Assessment, 2011. The UK National Ecosystem Assessment: synthesis of the key findings, in: UNEP-WCMC (Ed.), Cambridge.

UNEP-WCMC, 2011. Developing ecosystem services indicators: Experiences and lessons from sub global assessment and other initiatives, Technical Series. Secretariat of the Convention on Biological Diversity, Montréal.

van der Zee, D., 1990. The complex relationship between landscape and recreation. Landscape Ecol 4.

Vanpeene-Bruhier, S., Dalban-canassy, J., 2006. Etat des lieux de la connaissance et des attentes des acteurs sur l´ impact des infrastructures de transport terrestre sur les paysages et les écosystemes. CEMAGREF Unité de recherche des Ecosystemes Montagnard.

Willemen, L., Hein, L., Verburg, P.H., 2010. Evaluating the impact of regional development policies on future landscape services. Ecological Economics 69, 2244-2254.

Wilson, M.A., Hoehn, J.P., 2006. Valuing environmental goods and services using benefit transfer: The state-of-the art and science. Ecological Economics 60, 335-342.

Woodward, R.T., Wui, Y.-S., The economic value of wetland services: a meta analysis. 2001. Ecological Economics 37, 257-270.

Chapter 2

# Assessing the loss of global climate regulation service induced by transport infrastructure construction[1]

[1]This work has been published as: Tardieu, L., Roussel, S., Salles, J.-M., 2013. Assessing and mapping global climate regulation service loss induced by terrestrial transport infrastructure construction. Ecosystem Services 4, 73-81.

## Abstract

The purpose of this paper is to broaden the terrestrial transport infrastructure assessment process into the field of Ecosystem Services (ES), *i.e.*, the benefits people derive from ecosystems. Taking into account ES in an ex ante assessment of public infrastructure projects is of critical importance for the improvement of transportation decision-making tools, such as environmental impact assessment and cost-benefit analysis. For environmental impact assessment, the integration of an ES based approach opens the possibility of measuring a loss in ES supply (and its economic value); this provides a means of selecting among different possible pathways for the infrastructure. For cost-benefit analysis, since the ES loss induced by the selected pathway is expressed in monetary terms, it can be integrated as a standard social cost in the analysis, permitting a more efficient control of natural capital loss. We illustrate these points by assessing the loss of a global climate regulation service due to the soil tillage and sealing caused by a terrestrial transport infrastructure construction, using the example of a high-speed rail in Western France. We select different optional routes among the proposed routes and analyse which routes have the least impact on the global climate regulation service and its economic value.

## Résumé

L'objectif de cet article est d'introduire la prise en compte des Services Ecosystémiques (SE) dans le cadre des procédures d'évaluation des projets d'infrastructures de transport terrestres. La prise en compte des SE dans les outils d'évaluation de projet (études d'impact environnemental et bilan socio-économique) peut sensiblement améliorer les décisions publiques. Dans le cas des études d'impact, l'intégration de ces considérations peut permettre de mesurer la perte de services engendrée par chaque tracé d'infrastructure et d'intégrer ces pertes en tant que nouveau critère de choix de tracé. Concernant le bilan socio-économique, la perte de services exprimée en termes monétaires peut permettre de donner une indication quant à la perte sociale engendrée par le tracé final en matière de capital naturel. Nous illustrons ces points, dans un premier temps, à travers l'étude de la perte d'un service : la régulation du climat global. La perte de ce service est mesurée de deux manières : par la perte d'un stock de carbone libéré dans l'atmosphère suite au labour des sols, et par l'artificialisation des sols ne permettant plus la séquestration et le stockage du carbone. Nous étudions la perte en termes physique et économique dans le cas de la construction d'une ligne grande vitesse dans l'ouest de la France. L'étude est faite sur différentes options de tracés mais peut être applicable à l'ensemble des options de tracés.

## 2.1 Introduction

Terrestrial Transport Infrastructures are often considered as essential for economic development due to their contribution to time savings, comfort, safety, and regional accessibility, yet they have major impacts on the natural areas they cross. These impacts can involve direct, indirect and cumulative effects (Tricker, 2007). The conversion of natural areas into artificial areas, as a result of the construction, causes habitat loss and fragmentation with consequent declines in biological diversity (Quintero and Mathur, 2011). As a consequence, the compromise between social gains from infrastructure construction and the ecological and social losses induced by the environmental alteration requires analysis.

Recent improvements to environmental impact assessment of infrastructure construction projects provide much-needed guidance to public policies. In many countries, infrastructure projects are assessed regarding several criteria (flora, fauna, fragmentation, etc.) in order to avoid or minimize their environmental impact. However, and despite improvements to the process, the criteria used remain mostly qualitative. Moreover, the approach consists of weighting the different impacts with impact scores and assessing the overall impact by summing these scores (Geneletti, 2005). These scores are thus of critical importance, and as Geneletti (2006) argues, the process acts as if the scores have additive properties. In addition, at the present time, the loss of an Ecosystem Service (ES), *i.e.*, the benefits people derive from ecosystems, due to the construction is not quantified and is usually regarded as having little influence on the main infrastructure choices, such as time gains or the perceived economic viability of the project (Chevassus-au-Louis et al, 2009). The process of infrastructure projects' evaluation is usually performed through cost-benefit analysis. When cost-benefit analysis is used to enlighten decision-making for projects that impact the natural environment, monetary indicators of external effects have to be included in the assessment process for a greater efficiency.

Economists have developed a variety of methods that allow the construction of monetary indicators of non-market value loss associated with environmental and ecosystem impacts (TEEB, 2010). Taking into account ES in an ex ante assessment of public infrastructure projects is thus of primary importance if decision-making process associated with project selection is to be improved. Assessing ES changes and losses associated with an infrastructure projects can improve both (a) the process of choice for the least impact route for the transport infrastructure in terms of ES supply, demand and economic values in the environmental impact assessment, and (b) the integration of natural capital loss as a social cost in the cost-benefit analysis.

However, to our knowledge, there is only two studies which attempts to quantify the economic costs and benefits of infrastructure projects in terms of their impact on ES supply (Broekx et al, 2013 and SETRA, 2010). However, this studies lacks of spatial analysis and average economic values per hectare have been used for all services studied (only temperate forests and grasslands). Moreover, the studies retained the same impact area for all ES. In this paper, our objective is to broaden the scope of such project assessment to incorporate ES loss in order to provide for more efficient control of natural capital loss. To do so we assess the loss of a global climate regulation service associated with the destruction of habitats that contribute to carbon sequestration and storage by the construction of a high-speed railway in Western France. We select three optional routes among all the routes proposed in the discussed project, and analyse the loss in global climate regulation service and the economic value associated with each route. Studying the global climate regulation service allows us to avoid several methodological issues since the land-take[2] of the infrastructure on the service is reasonably well-known, and the marginal value of the damage is not modified by the loss amount. Obviously, other services will be impacted, and must be integrated in the analysis, but their measurement requires additional methodological advancement (*e.g.*, impact areas may exceed the area directly transformed by the transport infrastructure, consideration of beneficiaries and substitutes). Focusing on this service allows us to illustrate how, in first approach, the ES consideration can be conducted in this type of analysis.

The paper is organised as follows. In section 2.2 we quickly highlight the importance of evaluating ES loss due to the terrestrial transport infrastructure construction. In section 2.3 we describe the method used to assess and map the social loss of the global climate regulation service in order to select the route with least impact on this service (*ceteris paribus*) and its economic value. In section 2.4 we present our results which are discussed in section 2.5. Concluding remarks are formulated in section 2.6.

[2]The land-take is a hypothesis on a buffer that extends along the infrastructure axis where vegetation and land cover are supposed to be lost (Geneletti, 2006).

## 2.2 Transport infrastructure environmental externalities and ecosystem services loss

Transport infrastructure construction has increased rapidly in recent years, and continues to destroy and fragment natural ecosystems. In metropolitan France, the railroad network is currently about 30,000 km while the highway network now reaches roughly 1 Million km. In addition, public policies dedicated to planning and mobility involve a further 2,000 km of projected lines through 14 new high-speed rail projects before 2020.

Two assessment tools are used in transportation decision-making: Environmental Impact Assessment, intended to analyse and limit the impacts on the natural environment, and cost benefit analysis, intended to assess the benefits/costs ratio of the project. Terrestrial transport infrastructure projects involve a number of environmental externalities that alter ecosystem processes and functions and therefore ES supplied to human beings. The integration of ES assessment in the process could thus enhance the efficiency of both these tools.

### 2.2.1 Transport infrastructure impacts on ecosystem services

The effects of linear infrastructure construction on ecosystems and biodiversity are now well identified and can be classified in terms of either their direct or indirect impacts (Vanpeene-Bruhier and Dalban-Cassany, 2006). Direct impacts include all the losses of environmental features attributable directly to the infrastructure construction. This encompasses the loss of habitat and ecosystem area due to the conversion of the original land cover into an artificial surface (Geneletti, 2006). Indirect impacts include all the indirect losses of environmental features and processes induced by the interruption or the disturbance of ecological networks at different scales. Indirect impacts mostly involve (a) habitat fragmentation, *i.e.*, the break-up of natural areas into smaller and more isolated units which lose viability due to their small size (Geneletti, 2004), and (b) physical, thermal, visual or chemical barrier effects which can disrupt the flux of material and species within and between ecosystems and metapopulations (Vanpeene-Bruhier and Dalban-Cassany, 2006).

All these impacts can directly or indirectly affect ES supply. Direct impacts can disturb all types of ecosystem functions, bringing a total loss of ES in the area of influence of the infrastructure (provided that the impacts are not mitigated in the area). Indirect impacts are more complex and difficult to document; however it can be expected that they may mainly affect functions and processes related to species movements, habitat functions (lifecycle maintenance and gene pool protection), hunting recreation (deer, roe deer, and so on) or pollination services. Indirect impacts may also affect the scenic beauty of the landscape.

### 2.2.2 Integration of ecosystem Services in an environmental impact assessment

Environmental assessments practitioners have to consider increasingly ES in their assessment. However they lack of guidance on how to address ES, and thus their integration in the tool is still at its early stage and is rarely carried out explicitly (Landsberg et al., 2011; Geneletti, 2012; Honrado et al, 2013; Partidario and Gomez, 2013). Impact assessment key role consists in supporting the development of projects by assessing the environmental impacts that are likely to results from their construction. Integrating ES in the analysis would promote a more coherent assessment of environmental and socio-economics impacts; this would help to identify spatial and temporal trade-offs between humans and ecosystems (Geneletti, 2011). Following Geneletti (2012), Honrado et al. (2013), and Partidario and Gomez (2013), this integration requires in particular modeling ES in an explicit spatial manner.

Geographical information system is widely used as a supporting tool in various stages of the environmental assessment process (Atkinson and Canter, 2011). It is used mainly to: describe the baseline conditions of the project (hydrology, soils, topography, etc.), describe the impacts and predict their magnitude, assess the relative impact of alternative routes and thus the choice of the project with the lowest impact, and finally to identify areas where mitigation measures should be applied (Jao and Fonseca, 1996).

Analysis of the spatial dimensions, distribution and welfare associated with ES has only recently been considered (Heidkamp, 2008; Kozak et al, 2011). Because linear infrastructures change territorial configuration and biophysical conditions, they modify in an overlapping way the ES quantity and quality and their supply to human beings, the benefits people derive from these services, and the values people attach to these benefits. The initial conditions and the importance of the changes strongly depend on the location and the surrounding components of the ecosystems that provide the services (figure 2.1).

As a result, the spatial dimension of ES plays a critical role in the assessment of ES loss.

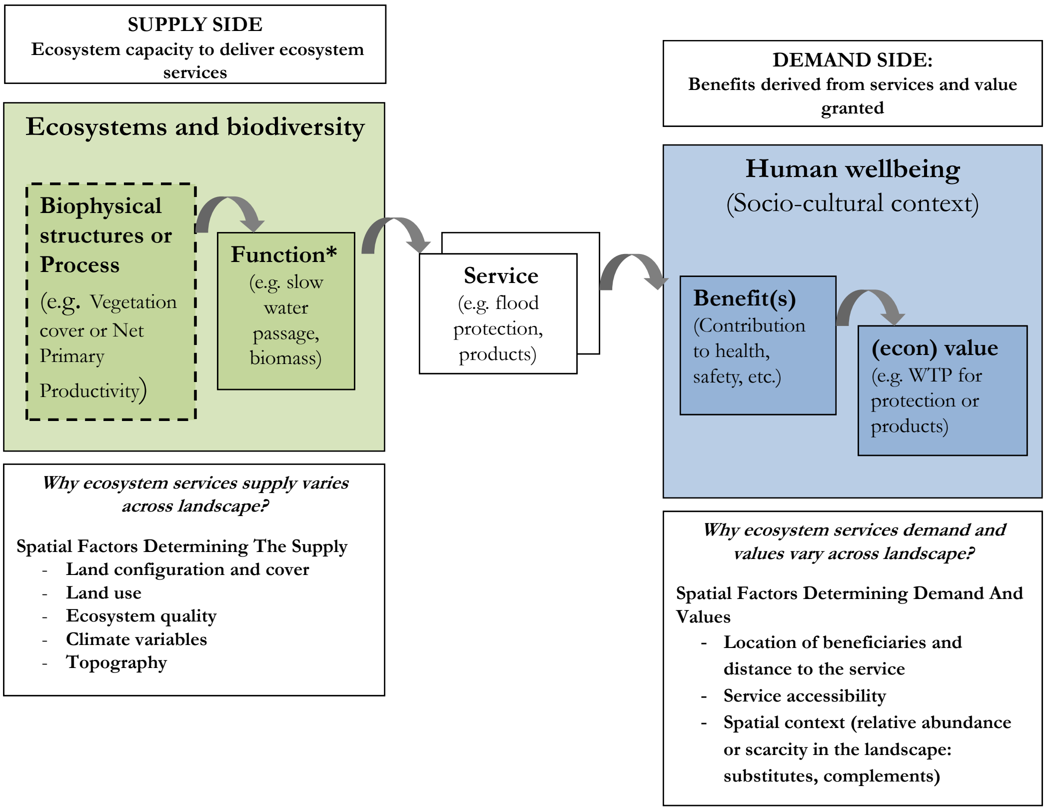



**Figure 2.1:** Spatial characteristics influencing ES supply side and demand side

While ES supply varies with habitat type, land use, land configuration, climate, hydrology, soil properties, fauna and topography (Burkhard et al, 2012; Nelson et al, 2009), ES demand varies in relation to the number of beneficiaries and their location, the socio-economic context, individual preferences and social practices (Boyd, 2008; Brander et al, 2012). Hence, as Brander et al (2012) argue, values depend on supply (quantity and quality of the available services), demand (number and socioeconomic characteristics of the beneficiaries), and on the spatial context of the underlying ecosystem providing the service (neighbor substitutes or complements).

The influence area of the infrastructure on ES is also of critical importance in the assessment of ES loss. It will depend on the ecosystem functions and services studied. Some services will be affected only by direct land-take of the infrastructure and others on a larger scale (especially if the service is related to a network feature).

### 2.2.3 Integration of ecosystem services assessment in cost-benefit analysis

Apart from the technical costs of the infrastructure, a number of environmental and social costs should be taken into account to more fully quantify the losses and gains generated by the infrastructure project. Since the official release of the French governmental report on investment choices in the transport sector (Boiteux et al, 2001), five types of non-market values are assessed in monetary terms in cost-benefit analysis of route selection: air pollution, greenhouse effect, noise disturbance, time gains, and the value of an enhancement of road safety. All these externalities are assessed for the usage of the infrastructure (the road traffic, etc.), but, there is currently no ex ante assessment of the losses induced by its construction and therefore by the habitat loss and the degradation of ecosystem functions it brings.

## 2.3 Assessing the global climate regulation service loss and its economic value: materials and methods

In this section, we examine the feasibility of assessing and mapping the ES values impacted by a infrastructure in a simple case study of high-speed rail infrastructure in Western France. The objective is to monitor an incremental change in ES supply due to the infrastructure, and then to assess and map changes in values compared to a baseline scenario without the infrastructure. The global climate regulation service includes carbon sequestration and storage in terrestrial ecosystems. This service has been increasingly emerging as one of the main issues in international climate negotiations (Polasky et al, 2011). Consequently, there is a great interest from policy-makers to get accurate information for this service in transport projects.

Beyond the atmospheric pollution induced by the fossil fuel combustion in the functioning infrastructure, which is now well assessed in the projects in France, Infrastructure construction implies land use change. $CO_2$ emissions through deforestation, biomass burning, wetlands drainage and soil tillage and sealing of natural ecosystems are very significant representing about 20% of the worldwide anthropogenic emissions (Lal, 2004). Land-use change also leads to service loss, impacting the future carbon sequestration and storage service by terrestrial ecosystems.

This global climate regulation service provides an illustrative example of ES integration in transport projects because it is:

1. A global service: carbon dioxide is a gas mixed in the atmosphere, and the global climate regulation benefits to all, wherever the beneficiaries are located;

2. Its loss is a function of direct habitat loss, according to the types of ecosystems; thus the impact area of the infrastructure will be essentially limited to the direct "land-take" of the infrastructure;

3. Its supply change is small relative to the global supply of the service, thus the marginal value will not change significantly with this loss.

### 2.3.1 Assessing the service supply change following the transport infrastructure implementation

Terrestrial ecosystems contribute to carbon sequestration and storage through several mechanisms. Carbon circulates permanently, in various chemical states, between the atmosphere, the biosphere, the lithosphere and the hydrosphere. Gaseous exchanges are thus made in the interface of various natural ecosystem types (forests, grasslands, oceans, wetlands, etc.). The potential for terrestrial carbon sequestration and storage varies across regions. Soil Organic Carbon (SOC) in metropolitan France is estimated to be 3 GtC. According to Arrouays et al (2002), three main factors determine the amount of carbon stored: local climate, topography and Land Use Land Cover (LULC). Thus, for a ground layer of 30 cm, carbon stocks are present in the grounds of forests or grasslands (70-80 tC/ha) (Arrouays et al, 2002). Carbon sequestration, *i.e.*, the vegetation capacity to fix atmospheric $CO_2$ by LULC depends on the occurrence of the LULC and if it has reached its storage equilibrium. Sequestration is a non-linear process, often greater in the first years of the LULC, thus there is a risk of overestimation, if the extrapolation of the average annual flow is performed in the long run.

To assess this ES supply, we make three technical assumptions. First, we assume that the actual stock is in equilibrium in order to avoid any carbon sequestration overestimate. Second, we assume that the implementation of the project causes the loss of the entire service in the area impacted by the infrastructure with a buffer zone of 100 m aside the axis (the construction implying the destruction of the vegetation and the soil tillage on a width of approximately 50 meters on each side of the axis). Finally, the loss is assumed to be fully effective in the first year of construction.

The natural carbon stored is identified by two variables.[3] The first is the Vegetation Carbon Stocks (VCS) contained in vegetation below and above ground biomass for each LULC. The second is the Soil Organic Carbon (SOC) contained in the soil organic matter of each soil by LULC (0-30 cm).[4] Carbon sequestration is ignored given the steady state assumption. With geo-referenced data it would have been possible to accurately determine the service loss; however, because usable local information was unavailable, we used regional and national data per LULC type. The VCS data come from a literature review, whereas the SOC data are provided by InfoSol GISSOL (INRA Orléans) (see Table 2.1 and Table 2.2 and Appendix A).

**Table 2.1:** References of data used to assess the service supply

| Data | Source | Input Data | Scale |
|---|---|---|---|
| SOC (0-30 cm) | Martin et al (2011), InfoSol GISSOL | LULC, Soil coverage, Climatic data (temperature, precipitation, potential evapotranspiration), Vegetation net primary production, Soil moisture regimes | National |
| VCS | IFN (2010); Lousteau et al (2004); IPCC (2006) | Climate, Vegetation net primary production, density of trees, biomass and height | Regional and continental region (temperate oceanic for Europe) data |

**Table 2.2:** SOC and VCS data used to assess the service supply

| Land use and cover types | SOC | VCS | Total in TC/ha | Total in $CO_2$/ha |
|---|---|---|---|---|
| Grassland/shrub | 85.2 | 6 | 91.2 | 334.43 |
| Perennial croplands | 45 | 32.6 | 77.2 | 284.56 |
| Annual croplands | 51.6 | 5 | 56.6 | 207.55 |
| Forest | 83.8 | 78 | 161.8 | 593.32 |
| Wetlands | 235 | 43 | 278 | 1,019 |

[3] Two other carbon pools can be considered (Conte et al, 2011): carbon stored in litter and dead wood and in the harvested wood products made with wood removed from the area (furniture, paper). But given the lack of local data we ignored both stocks.

[4] Changes in SOC levels are expected to be more rapid in topsoil (0-30 cm) than in deeper soil (JRC report, 2012).

### 2.3.2 Carbon economic value

For the economic value for the global climate regulation service, we use the opportunity cost of meeting mitigation policy goals in terms of costs and efficiency (Chevassus-au-Louis et al, 2009; Quinet et al, 2009). This method is considered more robust than the avoided damage cost method because it does not require making assumptions concerning any hypothetical future damage, and reduces the uncertainty of economic impacts of climate change (Tol, 2012). Moreover, in the CBA perspective of public investment choices, it seems acceptable to take as a reference the national commitment resulting from an international agreement.

In their analytical framework, Quinet et al (2009) defined a reference value of carbon to be used as a guideline for public investment choices in France. This method does not define any optimal mitigation goal, but assesses the carbon emission social opportunity cost of reaching a given mitigation goal on a given time horizon. Previously, the carbon value used for the investment choices in the transport sector was the one taken from the so-called Boiteux Report (2001). This study suggested a value of 27€/t$CO_2$ for the 2000-2010 period (that is 100€/tC[5]; for simplicity, we state our results in the rest of the paper in €/t$CO_2$.

The Quinet report defines the carbon value trajectory compatible with the fixed target, which corresponds to the European mitigation goals. These goals are to reduce greenhouse gas emissions by 20% in 2020, and by 60% to 80% in 2050 (which, according to the IPCC, would allow limiting the concentration at 450 ppm $CO_2$ equivalent). To identify the value trajectory inciting the economic agent to adapt and reach these objectives, Quinet et al (2009) used three simulation models[6] and obtained the following result (Figure 2.2). They start with the Boiteux value in 2010 and reach the pivot value of 100€/t$CO_2$ by 2030 (values in 2008 euros); this scenario deviates from a Hotelling rule but allows releasing the pressure on economic growth and facilitates the management of economic and social transitions. This gives for 2010 a value of 32€/t$CO_2$, 56€/t$CO_2$ for 2020 and 100€/t$CO_2$ for 2030 (namely a growth of the value at 5.8% per year). After 2030, a 100€/t$CO_2$ value

[5] 1 tC = 3,667 t$CO_2$.

[6] POLES (economic model representing agents behaviour, investments choice, and the energetic markets balance, simulation of the energy system in partial equilibrium), GEMINI-E3 (general equilibrium model, representing the macro-economic cost of the mitigation policy in terms of GDP variations or well-being), and IMACLIM-R (hybrid model of general equilibrium and detailed sector model). An optimal control model defined to estimate the emission optimal path of a limited carbon budget supplemented these models (Quinet et al, 2009).

increases following the Hotelling rule (then at the public discount rate 4%).

Under these assumptions, the carbon value increases from 100€/t$CO_2$ in 2030 to more than 200€/t$CO_2$ in 2050. Given the uncertainty in the long run (revision of previous commitments or greater technical progress), a value for 2050 is allowed to vary between 150€and 350€/t$CO_2$.

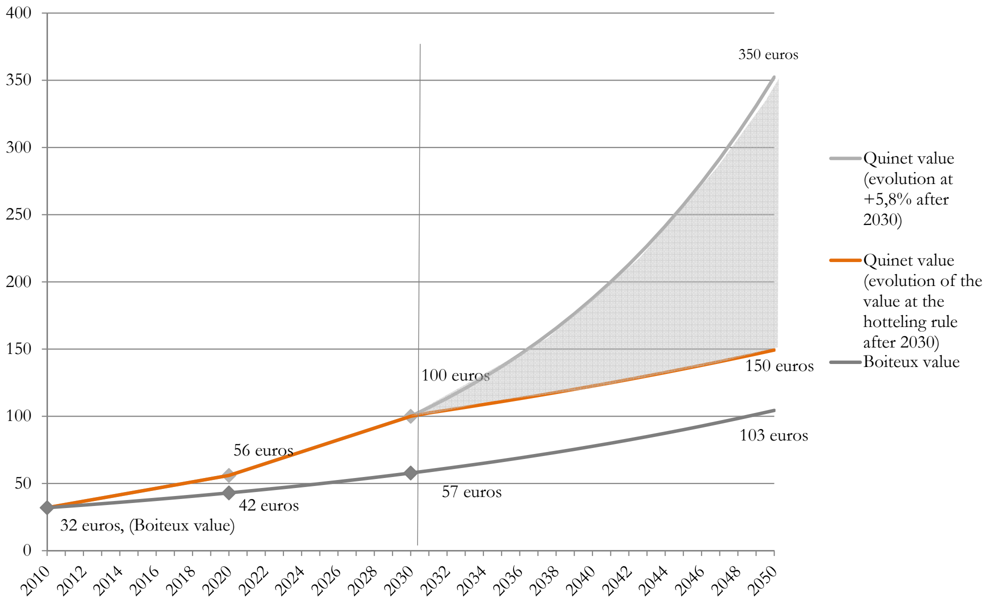



**Figure 2.2:** The Quinet and Boiteux value (in euro for the year 2008) adapted from Chevassus-au-Louis et al (2009)

### 2.3.3 The stock value and evolution parameters

We used two methods to assess the loss of carbon stocks. The first aims at calculating the cost of the carbon stock emissions. The calculation is then the simple multiplication of the stock loss, induced by the soil sealing, by the carbon value at the time of emission. The second approach aims at calculating what would have been gained if the stock had remained in the soil during the period instead of being released. The loss is then equal to what would have been gained if the infrastructure had not been built. In this case, we lose the storage service in addition to the cost of the stock released; keeping the stock in the soil contributes to delay the damages and the opportunity to release this stock later. Assessing this loss allows us to estimate the economic value granted on delaying a given damage.

Time discount rates vary among available studies: Stern (2007) proposed a very low 1.4% discounting rate, and until the Lebègue report (2005) the public discount rate was set at 8% for public investments in France. There has been an intense debate on the long-term discount rates during the last decade. In the case of the French public rate, two main reasons have led to the revision of this rate from 8% to 4%: firstly, the uncertainty on future economic growth which justifies a rate reduction; secondly, the emergence of environmental concerns towards future generations (climate change, biodiversity erosion, etc.) regarding sustainable development.

We used the public discount rate of 4% per year, since the assessments of transport infrastructures projects have to comply with public rules. The modification in relative prices increases the environmental cost to reach the Boiteux value of 100€/t$CO_2$ in 2030, and after 2030 it increases with the discount factor, following the Hotelling rule. This change is already included in the Quinet value. Finally, the value of the soil stock depends mainly on the discount rate for delaying the damages gained from this storage service. We followed the recommendations of the French governmental report on the economic approach of biodiversity (Chevassus-au-Louis et al, 2009) and set a return rate equal to the time discounting rate at 4%.

## 2.4 Results

### 2.4.1 Case Study

In this section, we assess and map the global climate regulation service impacted by the construction of an infrastructure. Our study considers different routes option proposed in the stage of option comparison in the project. We study two different zones in a project of high-speed rail infrastructure in Western France. We studied three route options in the Zone 1 and two optional routes in the Zone 2. At this stage of the project each route is assumed to have a 100m buffer zone, because the precise land-take of the infrastructure is not already known. The routes cross different types of natural habitats: forests, croplands (annual and perennial), wetlands and grassland/shrubs in different proportions (Figure 2.3 and Table 2.3).

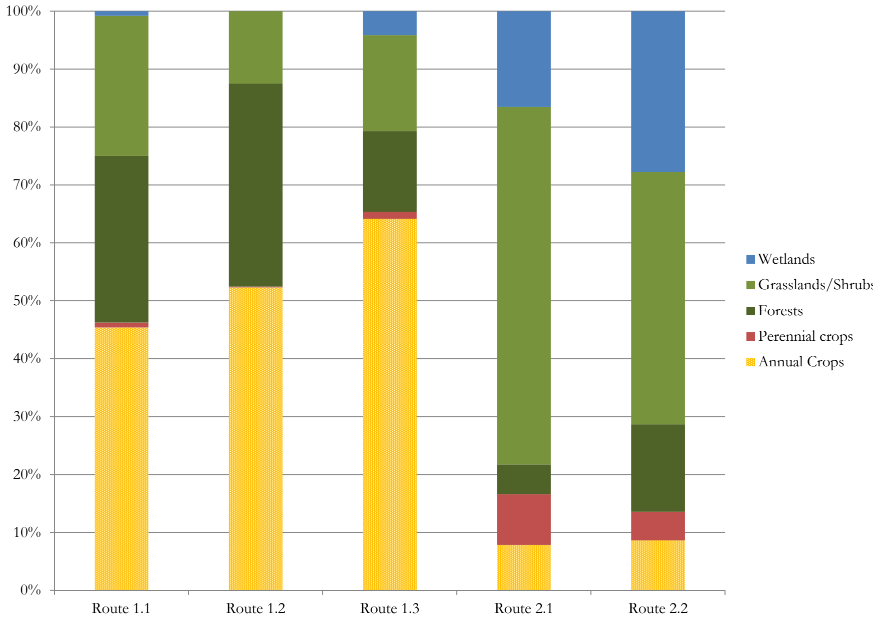



**Figure 2.3:** Ecosystems impacted by the different route options

**Table 2.3:** Habitats impacted for each alternative route

| LULC* | Description | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
|---|---|---|---|---|---|---|
| Grassland | Grassland areas, herbaceous areas and shrubs (ha) | 46.08 | 24.08 | 35.24 | 119.74 | 93.42 |
| Cropland | Cultivated crops (ha) | 86.50 | 100.78 | 136.38 | 15.23 | 18.52 |
| | Perennial crops (ha) | 1.63 | 0.30 | 2.55 | 16.98 | 10.59 |
| Forest | Areas dominated by trees (ha) | 54.81 | 67.51 | 29.55 | 9.93 | 32.35 |
| Wetland | Woody wetlands and herbaceous wetlands (ha) | 1.52 | 0.00 | 8.77 | 32.07 | 59.60 |
| | Total length (km) | 20.20 | 20.01 | 22.34 | 20.59 | 22.63 |
| | Total surface with buffer zone (ha) ** | 202.77 | 200.93 | 224.18 | 206.73 | 227.07 |

* Determined with Corine Land Cover classification 2006
** The difference with the sum of the area by habitat can be explained because some areas have no specific land use (*e.g.*, roads)

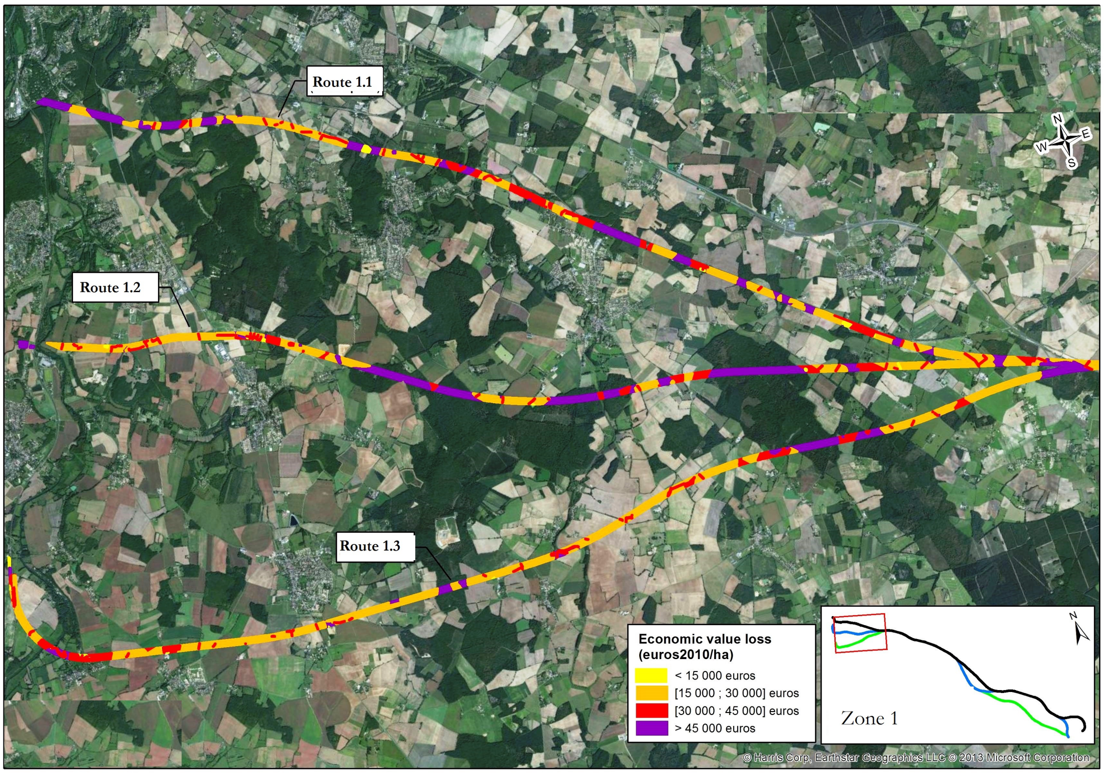
### 2.4.2 Service loss

We calculated the loss over a 55 year period (2015-2070, 5 years of construction plus 50 years of life expectancy of the infrastructure). For this time period, we assessed the value through the Quinet net present values and the interest placed every year for the storage service (Figure 2.3). The storage service value (SSV) is therefore given by the following equation:

$$SSV = \sum_{i=1}^{n}(PDD \times APV_i)$$

Where PDD is the Preference for Delaying a Damage (equal to the discount rate), and $APV_i$ is the Annual Present Value for year $i$, where $i = 1, ..., 55$ and $n = 55$.

Under these assumptions, the loss of the carbon storage service is about 112€/t$CO_2$ for 55 years. If we had considered that the storage service was lost in the very long term, the value would have varied from 112€to 165€/t$CO_2$. The 112€/t$CO_2$ monetary value is used to assess the carbon stock loss during the 2015-2070 time period, and map the loss of economic value for the ES for the three routes with the ArcGIS software. Figure 2.4 represent the infrastructure route options in Zone 1 and Zone 2 (bottom right of the maps). Five routes have been assessed here as an example but, when this type of calculation is performed on the actual project, many alternative routes cross each other. Consequently, choices have to be made on small route sections.



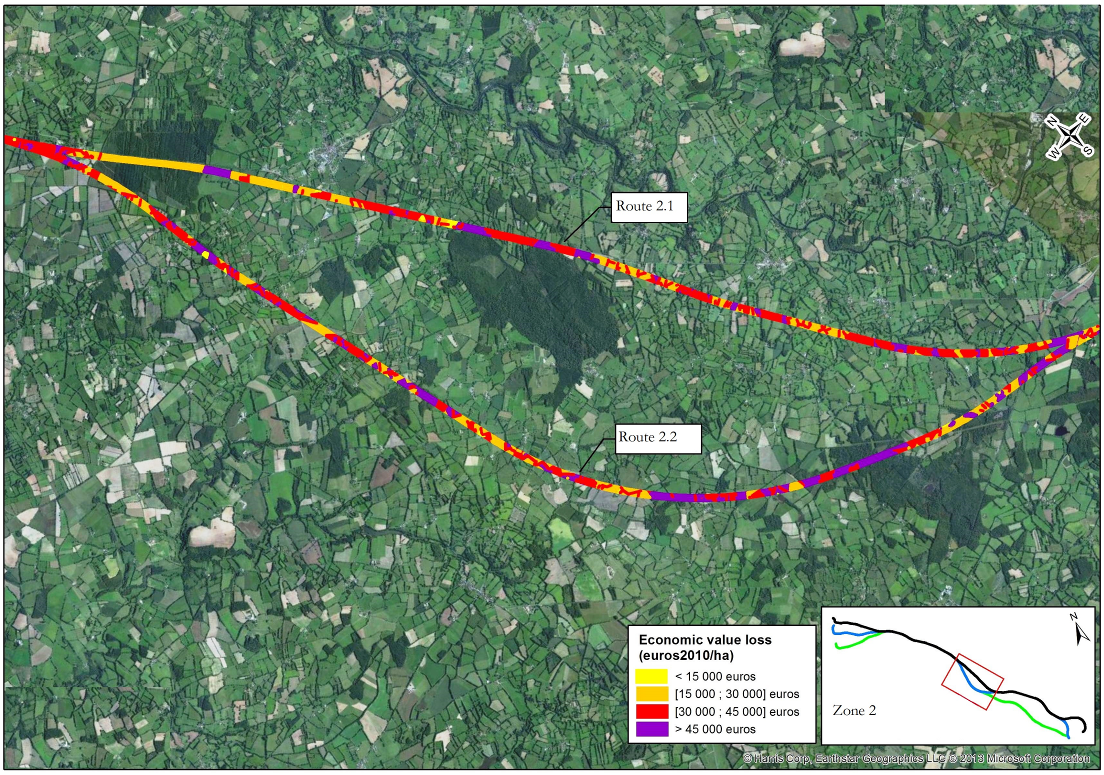




**Figure 2.4:** Economic value loss associated to the global climate regulation service for Zone 1 (above) and Zone 2 (below) alternative routes.

Furthermore, to represent the loss as a monetary cost in the CBA, we assessed each route under two options. The first method assesses the stock loss in the first year of construction assuming 44.7€/t$CO_2$ for the year 2013 (net present value in 2015). The second method assesses the storage service loss for 55 years assuming 112.1€/t$CO_2$ (Table 2.4).

**Table 2.4:** Global climate regulation service loss in euro for the year 2010 for each route option.

| **Option 1:** Value of the stock loss in the first year of construction | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
|---|---|---|---|---|---|
| Loss in tons of carbon dioxide ($CO_2$) | 67, 902 | 69, 113 | 67, 285 | 86, 607 | 118, 024 |
| $CO_2$ value per ton in 2015 | 44.7 | 44.7 | 44.7 | 44.7 | 44.7 |
| Loss in Million euros 2010 | 3.035 | 3.089 | 3.007 | 3.871 | 5.275 |

| **Option 2:** Value of the storage service loss for 55 years | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
|---|---|---|---|---|---|
| Loss in tons of carbon dioxide ($CO_2$) | 67, 902 | 69, 113 | 67, 285 | 86, 607 | 118, 024 |
| $CO_2$ value per ton for 55 years | 112.1 | 112.1 | 112.1 | 112.1 | 112.1 |
| Loss in Million euros 2010 | 7.611 | 7.747 | 7.542 | 9.708 | 13.230 |
| *Shaded cells highlight the least impacting routes* | | | | | |

The least impact option regarding the carbon storage service (ceteris paribus) is the Route 1.3 for Zone 1 and the route 2.1 for Zone 2, for both assessment options. The respective losses they imply are about of 3 M€ and 3.8 M€ under assessment option 1, and 7.5 M€ and 9.7 M€ under assessment option 2. The second assessment option multiplies by of about 2.5 the loss of the first assessment option for the service loss. The route 1.3 is the longest route, but it has the smallest impact on forest cover, conversely to route 1.2. Route 2.2 has a strong impact on natural wetlands which explains the loss it causes is much larger than the others route options.

## 2.5 Discussion

The conversion of natural habitats has been poorly assessed and valued in monetary terms in environmental impact assessment and CBA. As far as we know, the integration of criteria to assess loss in ES into such assessment has never been done for transport infrastructure, while $CO_2$ emissions from this type of change can be considerable. Jenny (1980) observed that:

> "among the causes held responsible for $CO_2$ enrichment, highest ranks are accorded to the continuing burning of fossil fuels and the cutting of forests. The contributions of soil organic matter appear underestimated"

And Lal (2004) noted:

> "There are no systematic estimates of the historic loss of Soil Organic Carbon (SOC) upon conversion from natural to managed ecosystems."

The case of total conversion is certainly the easiest and the more significant, especially in a country in which soil conversion is still a significant trend (Sainteny et al, 2011).

We believe that this analysis provides a meaningful level of the global climate regulation service. It is widely accepted that the idea of valuing ecosystem services on a global scale is probably unattainable and, beyond its initial provocative dimension, economically insignificant. This is why we consider it both more meaningful and useful to value the service based on the opportunity cost of meeting policy goals in terms of $CO_2$ emissions. It appears that this value is not very different to that obtained in the economic literature for the damage valuation perspective (Conte et al, 2011; Tol, 2009).

It is difficult to compare our work to existing literature because this type of assessment of the effect of terrestrial transport infrastructure construction on the storage service has not previously been performed. Nevertheless, we can compare this value with other land use change assessment, calculated with different methods. Our estimate of the carbon stock loss in the first year of construction is 44.7€/t$CO_2$. Compared for example to the results from Deng et al (2011), their value for one year is equal to 36.39€/t $CO_2$ for 2013 which is very close to our value even if the valuation method is different (relying on the carbon taxation value). Tol (2009) surveyed the peer reviewed literature on damage cost of carbon, and found a representative value is equal to 24€/t$CO_2$ for 2013. With the same evolution parameters we obtain a value for 55 years equal to 116€/t$CO_2$ for Deng et al (2011) and on average 98€/t$CO_2$ for the Tol (2009) value. We can also compare the stocks considered in the soil and above and below ground biomass. Our values of SOC stocks are on average 41% lower than the SOC values from multiple literature references reviewed in Polasky et al (2011). But they are 49% higher than the values presented in a French study (Arrouays et al, 2002).

Within our framework, the monetary valuation is not essential for the route selection in the EIA because the assessment of the service supply in physical terms would provide the same results when comparing the alternative routes. However, the second aim of this work was to integrate the economic value loss induced by the infrastructure construction in the CBA as a cost to be compared with other costs and benefits. We aimed at improving the project evaluation, and this analysis can be widened to any development project involving soil sealing by adjusting the infrastructure width.

Some limitations should be mentioned: a more precise mapping with sharp georeferenced data would have allowed us to take into account topography, soils type, and possibly other differentiation factors of carbon contents. This might improve the indicator for decision-making. The values on carbon stocks are probably underestimated because we do not take into account the carbon stored in dead wood and in harvested wood products. As for other monetary valuation method, several assumptions can be criticized. First, the economic valuation framework may be considered as inappropriate by decision makers that prefer to integrate other value paradigms (ecological, anthropological, etc.). Second, a high degree of uncertainty may hamper the consideration of the long term because of assumption made on the future technical progress which will allow reaching the $CO_2$ emissions reduction targets at lower economic and social costs than today. The future of the international agreements is also very uncertain. The result is also sensitive to the discount rate; our value of 4% reflects the discounting of an uncertain future economic growth. Besides, we implicitly admit that our work is done with acceptable marginal changes. Given all these uncertainties, the carbon value must be taken with precaution, and re-evaluation exercises can be set up to take into account the evolution of the parameters (Gollier, 2009; Quinet et al, 2009).

Our study allows for the introduction of a more general analysis regarding the overall carbon budget. We assess the loss induced by the construction *stricto sensu* of any transport infrastructure (*e.g.*, road, rail), but the future amount of carbon emitted from functioning infrastructure will be different if it concerns a road or a rail. A railroad will contribute to reduce emissions through the transfer of persons who usually use a car before the construction, and a road will generate supplementary traffic increasing emissions beyond the transfer effect (Martin and Point, 2012). This effect is already taken into account in the infrastructure CBA.

Finally, we note that economic assessments that aim at improving infrastructure investment choices should integrate the other services (beyond carbon storage) impacted by the infrastructure. But, for several of these services, new challenges would be added to those we discussed for the global climate regulation service. For the ecosystem capacity to deliver the services (the supply side), the loss will depend on whether the production of services by the ecosystem is proportional to the impacted surface. Ecosystems may need a minimum surface to provide a service or the service provided by a marginal unit of large size ecosystems may not be as important as the first hectares of such ecosystem (Barbier et al, 2008; Koch et al, 2009). The ES supply can, on the other hand, depend on the functioning of a network (as ecological networks for habitat services). This implies that fragmenting a network will involve a greater loss than impacting single and isolated habitats (Gaaff and Reinhard, 2012). As a result, considering values per hectare without accounting for ecological networks may lead to underestimate the services and the value related to a particular habitat. Finally, the analysis should integrate the ES demand: the number, the location and the socioeconomic characteristics of the beneficiaries should imply distance decay effects (Pate and Loomis, 1997; Bateman et al, 2006; Kozak et al, 2011).

## 2.6 Conclusion

The first aim of this paper was to identify some specific difficulties related to the inclusion of ES in infrastructure projects assessment. We focused on the assessment of the global climate regulation service supply in a real case study: a high-speed rail in Western France. We analysed different route options and identified which itineraries have the least impact on this service in terms of stock loss, but also in terms of storage service loss. By mapping the service, this approach may offer a new criterion for decision-making regarding the route selection in environmental impact assessment, allowing the choice on small scales and small route sections of the infrastructure. Moreover, since it is expressed in monetary terms, the loss of ES can be included in the of the infrastructure cost-benefit analysis. This allows decision-makers to compare these costs to other advantages, such as the generally decisive value of time gains.

Our approach completes current transport environmental measures whilst providing ES valuation. In France, as in other developed countries, according to variation in the legislation, transport infrastructure projects are realized following three tiered mitigation rules that aim to limit impacts on biological diversity: the so-called "avoidance, mitigation and compensation" rules with ecological trade-offs. The avoidance rule states that impacts and degradation of habitats have to be avoided, as much as (reasonably) possible. The mitigation rule refers to the minimization of the impacts that cannot be avoided by adapting the characteristics of the project to limit damages (establishing corridors and buffer zones). Finally, compensation refers to managing tradeoffs for residual impacts through actions that generate ecological benefit, at least equivalent to the losses induced by the infrastructure (creating protected areas, protecting species habitats, etc.). These rules aim at achieving a no net biodiversity loss goal when constructing infrastructure. In France, although the ecological compensation measures have been introduced since the Law on Protection of Nature in 1976, there is still no clear doctrine on the specifications that must be met with these measures. In practice, compensation measures proposed by the operators are usually calibrated in units of surface of ecosystems whose equivalency is assessed by the administration without explicit reference to carbon stocks or fluxes. It is therefore not possible to develop a judgment of general value. Our contribution is principally related to the "avoidance" tier following the three tiered rules by enhancing the route selection in integrating a new criterion of interest: ES and more precisely here the global climate regulation service loss. This would have to be extended to all other ecosystem services impacted.

## References


Arrouays, D., Balesdent, J., Germon, J.C., Jayet, P.A., Soussana, J.F., Stengel, P., 2002. Contribution à la lutte contre l'effet de serre. Stocker du carbone dans les sols agricoles de France?. Expertise scientifique collective. Synthese du rapport, 32p. (Online) available from: `/www.inra.fr/content/download/9337/122783/version/1/file/synthese-francais.pdfS.`

Atkinson, S.F., Canter, L.W., 2011. Assessing the cumulative effects of projects using geographic information systems. Environmental Impact Assessment Review 31, 457-464.

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo, G.M.E., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321-323.

Bateman, I.J., Day, B.H., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values: Welfare measures, distance decay and total WTP. Ecological Economics 60, 450-460.

Boiteux (Rapport du groupe d'experts présidé par), 2001. Transport : choix des investissements et coûts des nuisances. Commissariat Général du Plan, Paris.

Boyd, J., 2008. Location, location, location: the geography of ecosystem services Resources for the Future, 11-15.

Brander, L.M., Brauer, I., Gerdes, H., Ghermandi, A., Kuik, O., Markandya, A., Navrud, S., Nunes, P.A.L.D., Schaafsma, M., Vos, H., Wagtendonk, A., 2012. Using meta-analysis and GIS for value transfer and scaling up: Valuing climate change induced losses of European wetlands. Environmental and Resource

Economics.

Burkhard, B., Kroll, F., Müller, F., Windhorst, W., 2009. Landscapes´ Capacities to Provide Ecosystem Services - a Concept for Land-Cover Based Assessments. Landscape Online 15, 1-22.

Chevassus-au-Louis, B., Salles, J.M., Pujol, J.L., Bielsa, S., Martin, G., Richard, D., 2009. Approche économique de la biodiversité et des services liés aux écosystèmes : contribution à la décision publique. Documentation francaise, Paris. (online) available from :/http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/094000203/0000.pdfS.

Conte, M., Erick, N., Carney, K., Fissore, C., Olwero, N., Plantinga, A.J., Stanley, B., Ricketts, T., 2011. Terrestrial carbon sequestration and storage, in: Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E. (Eds.), Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, Oxford, UK.

Deng, S., Shi, Y., Jin, Y., Wang, L., 2011. A GIS-based approach for quantifying and mapping carbon sink and stock values of forest ecosystem: A case study. Energy Procedia 5, 1535-1545.

Gaaff, A., Reinhard, S., 2012. Incorporating the value of ecological networks into cost-benefit analysis to improve spatially explicit land-use planning. Ecological Economics 73, 66-74.

Geneletti, D., 2004. Using spatial indicators and value functions to assess ecosystem fragmentation caused by linear infrastructures. International Journal of Applied Earth Observation and Geoinformation 5, 1-15.

Geneletti, D., 2005. Multicriteria analysis to compare the impact of alternative road corridors: a case study in northern Italy. Impact Assessment and Project Appraisal 23, 135-146.

Geneletti, D., 2006. Some common shortcomings in the treatment of impacts of linear infrastructures on natural habitat. Environmental Impact Assessment Review 26, 257-267.

Geneletti, D., 2011. Reasons and options for integrating ecosystem services in strategic environmental assessment of spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management 7, 143-149.

Geneletti, D., 2013. Assessing the impact of alternative land-use zoning policies on future ecosystem services. Environmental Impact Assessment Review 40, 25-35.

Gollier C., 2009. Copenhague 2009 : incertitude et prix du carbone, in: J., T. (Ed.), Politique climatique : une nouvelle architecture internationale, Complément 1A. La Documentation française, Paris. (Online) available from: http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/094000513/0000.pdfS.

Haines-Young, R., Potschin, M., Kienast, F., 2011. Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. Ecological Indicators.

Heidkamp, C.P., 2008. A theoretical framework for a spatially conscious economic analysis of environmental issues. Geoforum 39, 62-75.

Honrado, J.P., Vieira, C., Soares, C., Monteiro, M.B., Marcos, B., Pereira, H.M., Partidário, M.R., 2013. Can we infer about ecosystem services from EIA and SEA practice? A framework for analysis and examples from Portugal. Environmental Impact Assessment Review 40, 14-24.

IFN (Inventaire National Forestier), 2010. Campagnes d´ inventaire 2006 à 2009. (Online) Available from: http://inventaire-forestier.ign.fr/spip/S.
IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories. Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan.

Jenny, H., 1980. The Soil Resource: Origin and Behavior, New York.

Joao, E., Fonseca, A., 1996. The role of GIS in improving environmental assessment effectiveness: theory vs. Practice. Impact Assess 14, 371-387.

JRC reference report, 2012. The state of soil in Europe. European Environment Agency, Italy. (Online) Available from: /http://ec.europa.eu/dgs/jrc/downloads/jrc_reference_report_2012_02_soil.pdfS.

Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M.E., Hacker, S.D., Granek, E.F., Primavera, J., Muthiga, N., Polasky, S., Halpern, B.S., Kennedy, C.J., Kappel, C.V., Wolanski, E., 2009. Non-linearity in ecosystem services : temporal and spatial variability in coastal protection. Frontiers in Ecology and Environment 7, 29-37.

Kozak, J., Lant, C., Shaikh, S., Wang, G., 2011. The geography of ecosystem service value: The case of the Des Plaines and Cache River wetlands, Illinois. Applied Geography 31, 303-311.

Lal, R., 2004. Soil Carbon Sequestration Impacts on Global Climate Change and Food Security Science 304, 1623-1627. Landsberg, F., Ozment, S., Stickler, M., Henninger, N., Treweek, O., Venn, J., Mock, G., 2011. Ecosystem Services Review for Impact Assessment: Introduction and Guide to Scoping, WRI Working Paper. World Resources Institute, Washington DC.

Lebègue, D., Baumstark, L., Hirtzman, P., 2005. Révision du taux d´ actualisation des investissements publics. La Documentation Française, Paris. (Online) Available from `/http://catalogue.polytechnique.fr/site.php?id=324&fileid=2389S.`

Loustau, D.C., 2004. Séquestration de Carbone dans les grands écosystèmes forestiers en France. Quantification, spatialisation, vulnérabilité et impacts de différents scénarios climatiques et sylvicoles. . Rapport final du projet de recherche CARBOFOR, programme GICC 2001. (Online) Available from: `/http://www.gip-ecofor.org/doc/drupal/gicc/7-01LousteauCraboforRF.pdfS.`

Martin, J.-C., Point, P., 2012. Road project opportunity costs subject to a regional constraint on greenhouse gas emissions. Journal of Environmental Management 112, 292-303.

Martin, M.P., Wattenbach, M., Smith, P., Meersmans, J., Jolivet, C., Boulonne, L., Arrouays, D., 2011. Spatial distribution of soil organic carbon stocks in France. Biogeosciences Discuss 7, 8409-8443.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7, 4-11.

Partidario, M.R., Gomes, R.C., 2013. Ecosystem services inclusive strategic environmental assessment. Environmental Impact Assessment Review 40, 36-46.

Pate, J., Loomis, J.B., 1997. The effect on willingness to pay values : a case study on wetlands and salmon in california. Ecological Economics 20, 199-207.

Polasky, S., Nelson, E., Pennington, D., Johnson, K., 2011. The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. Environmental and Resource Economics 48, 219-242.

Quinet, A., Baumstark, L., Célestin-Urbain, J., Pouliquen, H., Auverlot, D., Raynard, C., 2009. La valeur tutélaire du carbone. Documentation française, Paris. (online) Availaible from: `http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/094000195/0000.pdfS.`

Quintero, J.D., Mathur, A., 2011. Biodiversity Offsets and Infrastructure. Conservation Biology 25, 1121-1123.

Sainteny, G., Salles, J.M., Duboucher, P., Ducos, G., Markus, V., Paul, E., Auvelot, D., Pujol, J.L., 2012. Les aides publiques dommageables à la biodiversité, in: Française, L.D. (Ed.). Centre d´ Analyse Stratégique. Paris. (Online) Available from: `/http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/124000434/0000.pdfS.`

SETRA, 2010. Monétarisation des externalités environnementales. Rapport d´ études, MEEDDM, 132 p. (Online) Available from: `/http://www.setra.equipement.gouv.fr/IMG/pdf/1017w_Rapport_monetarisation_externalites_environnementales.pdfS.`

Stern, N., 2007. The Economics of Climate Change: The Stern Review. Cambridge University Press, Cambridge.

The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. In: Kumar, P. (Ed.), Earthscan, London, Washington.

Tol, R.S.J., 2009. The Economic Effects of Climate Change. Journal of Economic Perspectives 23, 29-51.

Tricker, R.C., 2007. Assessing cumulative environmental effects from major public transport projects. Transport Policy 14, 293-305.

Vanpeene-Bruhier, S., Dalban-canassy, J., 2006. Etat des lieux de la connaissance et des attentes des acteurs sur l´ impact des infrastructures de transport terrestre sur les paysages et les écosystèmes. CEMAGREF Unité de recherche des Ecosystémes Montagnard, 114 p. (Online) Available from: /http://www.ittecop.fr/doc/genese/Cemagref.pdfS.

# Chapter 3

# Combining direct and indirect impacts on ecosystem service loss associated with infrastructure construction[1]

[1]A preliminary version of this work has been accepted for publication in Revue d'Économie Politique. The full version of the paper has been submitted as: Tardieu, L., Roussel, S., Thompson, J. D., Labarraque, D., Salles, J. M. Combining direct and indirect impacts on ecosystem service loss associated with infrastructure construction.

## Abstract

The destruction of natural habitats and the associated loss of Ecosystem Services (ES) are rarely jointly assessed and quantified in Environmental Impact Assessment (EIA). The objective of this chapter is to broaden the scope of terrestrial transport infrastructure project assessment by incorporating ES loss in a way that could enable more precise identification of ES loss. This is achieved by combining direct loss of ES with indirect loss due to impacts on landscape connectivity. For both direct and indirect effects we also integrate potential threshold effects on ES loss. More precisely, we first quantify how much of each type of ES is generated by different land units in the absence of the infrastructure (baseline conditions). We then estimate ES loss caused by infrastructure construction in a way that discriminates among different types of ES, some of which show losses that are directly proportional to the surface impacted and others which show additional indirect losses associated with landscape connectivity. In addition, we propose a method for assessing threshold effects in particular ecosystem types that are most sensitive to their occurrence. Based on the examination of part of a contemporary infrastructure project, we compare implementation options to provide an example of how choices can be improved by mapping ES loss associated with a combination of direct and indirect impacts. This kind of analysis could be used more generally to assess development projects simply by adjusting the framework for analysis in relation to the type of project and the ecosystems concerned.

## Résumé

La perte de Services Ecosystémiques (SE) associée à l'artificialisation d'habitats naturels est rarement évaluée et quantifiée dans les études d'impact environnemental. L'objectif de ce chapitre est d'élargir la portée de l'évaluation des projets d'infrastructures de transport terrestre linéaire en intégrant la perte de SE de manière à permettre une identification plus précise de la perte subie. Ceci est obtenu gâce à la combinaison de la perte directe et de la perte indirecte de services due aux impacts de l'infrastructure sur la connectivité des entités spatiales. Pour les deux types d'impacts nous intégrons des seuils potentiels sur la fourniture de services en proposant une méthode pour prendre en compte ces effets sur des écosystèmes particulièrement sensibles. Plus précisément, nous montrons comment quantifier dans un premier temps la fourniture de services par différentes unités spatiales en l'absence d'infrastructures (état initial). Nous estimons par la suite la perte causée par la construction de l'infrastructure de maniére à différencier, selon le type de service impacté et le type d'écosystème, les pertes directement proportionnelles à la surface impactée et celles montrant une perte supplémentaire associée à l'impact sur la connectivité des milieux naturels. Nous appliquons cette méthode à un cas de projet d'infrastructure de transport terrestre linéaire, et comparons différentes options de tracé afin de donner un exemple de la manière dont les choix pourraient être améliorés en cartographiant les pertes directe et indirecte de SE. Ce type d'analyse pourrait être utilisé plus généralement dans l'évaluation des projets de d'aménagements,

en ajustant simplement le cadre méthodologique en fonction du projet et du type d'écosystème concerné.

## 3.1 Introduction

Ecosystem Services (ES) are derived from the ecological functioning of natural ecosystems and are typically conceptualized as flows of goods and services that benefit human societies (Bagstad, 2009; Daly and Farley, 2003). Land use change associated with human population growth, urban sprawl and land development during the 20th century continues to alter and destroy natural ecosystems, with consequent degradation of ecological processes and an ever-increasing human footprint on natural ecosystems across the Globe (Millennium ecosystem assessment, 2005; Sala et al, 2000; Sanderson et al, 2002). There is thus increasing concern on how the impacts of such activities on ecosystem function affect the capacity of ecosystems to provide ES to societies (Broekx et al, 2013; Geneletti, 2013; Honrado et al, 2013; Kumar et al, 2013; Partidario et al, 2013).

The impacts of land conversion related to development projects, and here to linear infrastructure construction, primarily concern a reduction of the surface areas of natural ecosystems, due to their conversion into an artificial surface (Fahrig 2002; Geneletti, 2006), with subsequent ES loss (Gascoigne et al, 2011; Kreuter et al, 2001; Mendoza-Gonzalez et al, 2012). However, although integrating ES loss associated with development projects is a critical element in the improvement of Environmental impacts assessments (EIA), it is nevertheless a complicated task that requires careful attention. It is now clear that the area directly taken for roads and railways is, not a reliable measure of the loss of natural habitat. The disturbance influence of infrastructures and other development projects on surrounding wildlife, vegetation, hydrology, and landscape go well beyond the area that is physically occupied and can cause greater overall loss and degradation than that incurred in the zone of presence of the infrastructure (Trocmé et al, 2002). ES loss will thus depend on the type of ecosystem, the spatial extent of impacts on different ecosystems and how impacts affect spatial interactions among ecosystems and their components.

As Mitchell et al (2013) and Ng et al (2013) have recently discussed, landscape composition (the spatial cover of land use types in a given area) and configuration (the pattern of different elements in a landscape) can both affect the provision of ES. Indeed, indirect effects associated with modifications to landscape connectivity, *i.e.* the flow and movement of materials and organisms across a landscape (Taylor et al, 1993) may impact on a range of different ES (Table 3.1), making it possible that the destruction of an ecosystem in an impact zone may cause a decline in an ES provision by a landscape element exterior to the zone. Hence, ES loss will depend on impacts to the ecological interdependencies among the different elements of a landscape or territory (Thompson et al, 2011; Ng et al, 2013). According to Mitchell et al (2013), neglecting this landscape connectivity can lead to a failure to properly quantify impacts on ecosystem services. Indeed, disrupting landscape connectivity can have as much effect on ES as direct impacts on the surface of a particular ecosystem in the impact zone (Gilbert-Norton et al, 2010; Mancebo-Quintana et al, 2010). The challenge here is thus to provide methods that integrate the differential response of ecosystems to loss in spatial cover and the indirect impacts associated with loss in landscape connectivity.

**Table 3.1:** Indirect effects on ecosystem function that may translate into ecosystem service loss beyond a direct impact zone due to landscape connectivity and interactions across ecological systems

| Ecosystem services | Ecological function | Reference |
|---|---|---|
| Freshwater supply | Nutrient inputs and water flow from neighbouring terrestrial ecosystems impact on water quality of rivers and estuarine ecosystems | Turner and Rabalais (2003)<br>Brauman et al (2007)<br>Lalande (2013)<br>Barbier et al (2011)<br>King et al (2005) |
| Water flow and flood regulation | Disruption of river flows, sediment movement and deposition | Arthington et al (2009)<br>Opperman et al (2010)<br>Brauman et al (2007) |
| Pollination | Pollination of agricultural crops by insects that require natural areas around areas of cropland loss. Here, land around the area providing the ecosystem service, and not just the crop land itself, provides the service because of movement of insect pollinators, herbivores, and predators from patches of natural habitat to adjacent agricultural fields | Tscharntke and Brandl (2004)<br>Kremen et al (2002)<br>Nicholls et al (2001)<br>Brosi et al (2008)<br>Carré et al (2009) |
| Recreation | Importance of landscape diversity, impacts on particular elements of a landscape that alter the whole landscape | van der Zee (1990) |
| Hunting | Mammal movements disrupted by linear infrastructures and other factors associated with habitat fragmentation | Fahrig and Rytwinski (2009)<br>Beier (1995)<br>Hilty et al (2006) |
| Fishing | Commercial fisheries influenced by the connectivity of coastal marine ecosystems, freshwater fish populations depend on river connectivity | Meynecke et al (2008)<br>Holmlund and Hammer (1999) |

In addition, species and ecosystems often need a minimum surface area to complete their ecological functions. Hence, when part of a habitat or an ecosystem is lost due to development, species responses and ecosystem function may show a non-linear response to land conversion due to threshold behaviour (Groffman et al, 2006; Lindenmayer and Luck, 2005; Muradian, 2001; Swift and Hannon, 2010). Such threshold behaviour is observed when the response of an ecological factor (individual behaviour, abundance of a species, community composition, ecosystem flows) shows a marked change in relation to the amount of habitat in the landscape once it falls below a certain level. For example, the abundance of a species in a landscape may decrease more steeply with habitat loss once the amount of remaining habitat falls below some proportion of the total landscape area (Swift and Hannon, 2010). This proportion represents the threshold. Also, in terms of landscape connectivity, a particular habitat patch may play a key role in movement patterns (Gaaff and Reinhard, 2012), particularly for large mammals (Forman and Deblinger, 1999; Hilty et al, 2006). Its loss may thus incur a threshold effect. Likewise for recreation, the loss of a particular unique landscape element may have a large effect relative to its surface area.

The occurrence of such thresholds will depend on the biology of the species (dispersal traits, reproduction, and demography), the degree of habitat and ecosystem fragmentation across the landscape, and the scale of ecosystem function (Poiani et al, 2000; Swift and Hannon, 2010). As Poiani et al (2000) illustrate, for ecosystems with a highly localised scale of function, even a small loss in the surface area is likely to cause a large loss in ecosystem function. Since threshold levels are expected to vary by species, landscape type, and spatial scale, the results of one study do not necessarily apply to another situation (Huggett, 2005; Lindenmayer and Luck, 2005; Lindenmayer et al, 2005), making it difficult to integrate this concept into management decisions (Groffman et al, 2006; Sudding and Hobbs, 2009). It is however clear that assessing variation among ecosystems in their sensitivity to threshold behaviour is a major challenge to our efforts to assess ES loss associated with development projects.

In this chapter, our overall objective is to test the feasibility of assessing ES loss involved by different implementation options of a major linear infrastructure development. Our applied framework focuses on the comparison of the ES loss for different route options for a high-speed rail project in France[2]. In addition to accounting for direct losses of ES provision due to the conversion of ecosystems, we integrate the possibility of indirect losses

[2]We cannot disclose information on the case study due to contractual commitments with *EGIS Structures et Environment.*

associated with landscape connectivity. Furthermore, we provide a preliminary attempt to assess potentially non-linear responses due to threshold effects. This allows us to provide a broader and more comprehensive assessment of ES loss both in terms of biophysical quantities and economic values. The ES loss assessment we propose can be adapted to the assessment of different types of linear infrastructures (highways, railways) by adapting land takes to the local ecological and landscape context.

In Section 3.2, we present the methodological options we take in order to assess direct and indirect ES loss. In Section 3.3, we present the results in our case study of high speed rail implementation, and the results are discussed in Section 3.4. Finally, Section 3.5 concludes.

## 3.2 Methodological options and data collection

The general framework for analysis is displayed in Figure 3.1 and the valuation steps adapted to the infrastructure construction are described in the following sub-sections.

### 3.2.1 Identification of ecosystem services supplied by the study area

The definition of the study area, *i.e.*, the area potentially affected by the presence of the project, must have a boundary sufficiently large around the axis of the route to take into account indirect effects operating at a larger scale. In the preliminary phase of the project, the environmental analysis is made to find contrasting route options. In this way, the study area is large enough to allow studying key effects, which will occur beyond the direct impact of the project. The use of an accurate land-use/land-cover (LULC) typology in the study area is an important step (Bagstad, 2009). It depends on the geographical data available for the study site. Here, we use LULC data using the Corine Biotope typology (with a scale 1/5,000).

For the classification of ES, we use a classification similar to the one described in the first chapter of the TEEB report (2010). It is difficult to be totally exhaustive; hence, we could not include some significant use values derived from freshwater provisioning, water treatment and aesthetic information. Still, these services are not totally overlooked in the analysis, as regulatory incidences assessments are required in the EIA framework[3]. We also

[3]For Freshwater services the French Water Law requires a detailed description of the impacts on aquatic areas, and a description of mitigation and maintenance measures. For landscapes and aesthetic information,

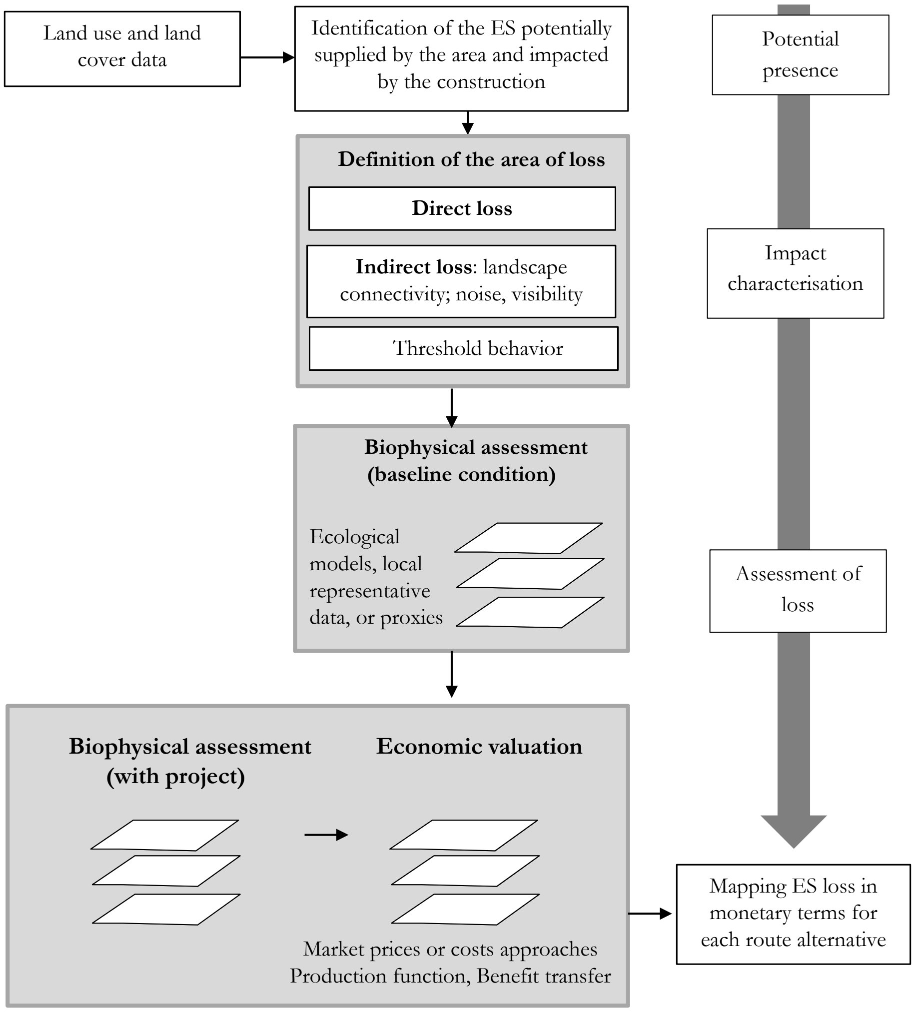



**Figure 3.1:** Methodological framework summary

ignored habitat services and soil formation, considered as support services, to avoid double counting (for instance the habitat service can be seen as a support service supporting the required conditions for wild game flora and fauna to develop, hence the benefit people service from this function is the use through recreation features). Finally, given that we consider the net social loss, we took into account only the ES that are not compensated at other moments in the evaluation process. Hence, we did not consider the marketed ES related to agricultural and forestry production whose losses are already taken into account as a cost (to the producers and landowners) in the project[4]. Instead, food and timber

the law on rural territory development requires an evaluation of the impact on landscape, and the prevision of landscaping actions to restore the features. Environmental vulnerability regarding these services are thus already mapped and taken into account for the implantation option choice.

[4]Compensation payments are paid to farmers and landowners in order to cover the loss of income (on the basis of the average marginal gains of the culture), loss of fertilization, loss of productivity of remaining cultures for farmers and the loss of the land value (based on land market values), the land opportunity

provisioning services considered are only those associated with natural products (berries, mushrooms) and firewood.

Inference of ES from LULC typology was made in two ways: first, based on literature that links Corine Land Cover (2006) to ES (Burkhard et al, 2009; CGDD, 2010) and second, from studies that document a particular relationship between a land cover type and an ES, *e.g.*, flood protection provided by inland natural wetlands. The typology developed for the study is presented in Figure 3.2.

To be considered as an ES, ecosystems should perform functions that benefit human societies. Whether or not a service is provided by an ecosystem or a type of land-use will depend on its spatial characteristics and location, hence our classification integrates a notion of conditionality on supply and demand based on the location of an ecosystem (Figure 3.2). Presence can be directly deduced from ecological modeling of the service, however some services are not assessed through models and so assumptions of ES conditional presence have to be explicitly defined. For instance, the flood protection service is considered as being supplied only by forests and wetlands present in the floodplain of a river threatening an inhabited zone (which itself benefits from the service); whereas the recreation service is considered as being demanded only by ecosystems sufficiently close to an urban area (Eade and Moran, 1994).

### 3.2.2 Characterisation of impacts on ecosystem services

Linear infrastructure construction can directly or indirectly affect ES supply. Identifying the terrestrial transport infrastructure influence area on ES is thus complex. Whether or not we consider that the ES loss is partial or total after the infrastructure construction, it is necessary to determine the surface on which this loss will occur. For some ES, the loss can simply be assessed in relation to the surface directly covered by the infrastructure, and for others, due to an impact on landscape connectivity the loss should be assessed for a surface that is greater than the surface associated with the direct impact, *i.e.*, goes beyond the width of the infrastructure. This integration of indirect impacts is similar to the analysis proposed by Forman and Deblinger (2000) who determined an "effect zone" which is the area over which significant ecological effects extend beyond the infrastructure axis (see also : Trocmé et al, 2002). We thus constructed a typology of ES loss subsequent

cost, and loss of the value of nearby lands for landowners.

Classification of ES supplied and impacted for each land cover type in relation to their potential (●) or conditional (○) presence or effects. Rows represent the land-cover types present in the different route options and columns represent the ES potentially present and potentially impacted by the project. Unshaded combinations of land cover types and ES represent those that incur only a direct loss. Shaded combinations represent those which incur indirect losses, over and above those due to a direct loss, as a result of a disruption in landscape connectivity (see Table 1), which in the case of recreation (*) involves an impact on a particular point of interest.

| | | Provision | | | Regulation | | | | | | | | | Cultural | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *Land cover type* | Wild Food | Raw materials | Freshwater | Air quality | Local climate | Global Climate | Flood protection | Water flow | Waste treatment | Erosion prevention | Pollination | Biological control | Recreation * | *Hunting* | *Fishing* |
| Non Marine Water | Water bodies | | | ● | | | | | ● | ● | | | | ○ | ● | ○ |
| Grassland | Sclerophyllous vegetation | | | ● | | | ● | | | ● | ○ | ○ | ● | ○ | ● | |
| | Moors and heathlands | | ● | ● | | | ● | | | ● | ○ | ○ | ● | ○ | ● | |
| | Transitional woodland shrubs | | ● | ● | | | ● | | | ● | ○ | ○ | ● | ○ | ● | |
| | Natural grassland | | ● | ● | | | ● | | | ● | ○ | ○ | ● | ○ | ● | |
| Forest | Broad-leaved forest | ● | ● | ● | ○ | ○ | ● | ○ | | ● | ○ | ○ | ● | ○ | ● | |
| | Coniferous forest | ● | ● | ● | ○ | ○ | ● | ○ | | ● | ○ | ○ | ● | ○ | ● | |
| | Mixed forest | ● | ● | ● | ○ | ○ | ● | ○ | | ● | ○ | ○ | ● | ○ | ● | |
| Wetland | Alluvial forests and thickets | | ● | ● | ○ | ○ | ● | ○ | ○ | ● | ○ | ○ | ● | ○ | ● | ○ |
| | Inland marshes | | ● | ● | | | ● | ○ | ○ | ● | ○ | ○ | ● | ○ | ● | ○ |
| | Peatbogs | | ● | ● | | | ● | ○ | ○ | ● | ○ | ○ | ● | ○ | ● | ○ |
| | Wet grasslands | | ● | ● | | | ● | ○ | ○ | ● | ○ | ○ | ● | ○ | ● | ○ |
| Cropland | Pastures | | | | | | ● | | | | ○ | | | ○ | ● | |
| | Annual and permanent crops | | | | | | ● | | | | ○ | | | ○ | ● | |
| | Fruit trees, olive groves, vineyards | | | | | | ● | | | | ○ | | ● | ○ | ● | |
| Hedgerow | Screens trees and hedges | ● | ● | ● | | ○ | ● | | | ● | ○ | | | | ● | |

Attribution of conditional presence for ES provision and demand (○) was based on the following: *Air quality and local climate regulation*: only for urban or periurban areas; and for hedgerows situated on the edge of a culture. *Flood protection*: land cover types located in the floodplain of a river. *Regulation of water flows*: only if the wetland is related to a river system. *Erosion prevention*: depending on the slope, rainfall erosivity, soil erodibility, conservation practices and vegetation retention efficiency. *Pollination*: for entomophiloius crops needing insect pollination. *Fishing*: for wetlands adjacent to a river. *Recreation*: for natural areas situated in the proximity to a city.

**Figure 3.2:** Classification of ES supplied and impacted for each land cover type

to infrastructure construction which integrates direct losses and indirect losses related to landscape connectivity (Table 3.1 and Figure 3.2).

We also provide a preliminary attempt to integrate threshold effects to evaluate how they may potentially affect the outcome of ES provisioning. Although the concept of ecological thresholds has generated much interest (see introduction), the identification of threshold behaviour in natural systems and in particular the precise threshold levels that push ecological systems onto different trajectories, is limited. Most evidence (see review by Swift and Hannon, 2010) concerns threshold responses in terms of species loss from fragmented animal communities which can show threshold behaviour once the cover of a given habitat declines below 10 to 30% land cover in some cases (Andrén, 1995) or even below 50% in species whose movement may incur high mortality rates (Flather and Bevers, 2002). Some ecosystems, with a highly localised scale of function, may be particularly prone to threshold effects, and even a small initial loss in the surface area of the ecosystem may cause a large loss in ecosystem function. This is particularly the case for ecosytems with a localised scale of functioning such as inland marshes, water bodies and peat bogs which mediate water flow regulation and quality (Poiani et al, 2000; Muradian, 2001). Indeed, freshwater regulation may be more generally affected by threshold effects associated with land use transformations (King et al, 2005).

To integrate potential threshold behaviour into our analysis, two types of loss are considered. For some ecosystems, *i.e.*, inland marshes, water bodies and peat bogs, we considered a total loss of services such as the provision of freshwater, the regulation of water flow, flood prevention and waste treatment regardless of the area of these ecosystems that is directly impacted. This can be justified by the fact that infrastructure construction on such areas will involve wetland drainage and evidence of an greatly extended effect of this type of impact has been demonstrated (Forman and Deblinger, 2000; Findlay and Bourdages, 2000; Seiler, 2002). For other ecosystem types that furnish these services, but which probably function on a larger scale, we considered that once 50% of the ecosystem surface is impacted then a threshold behavior may occur with a total loss of ES provision. This ratio of change can be modified according to the characteristics of ecosystems present in a given study area. A representation of this effect is given in Figure 3.3.

Recreation is a particular case, because the service is not directly connected to an ecological function. Indirect loss can occur due to the integrity of the landscape which is lost even when only a small area is impacted at a particular point of interest. In addition, the infrastructure may cause a loss of landscape interest through the infrastructure visi-


*The different patches of wet grasslands are considered as impacted in different ways: the patch can be totally impacted because more than 50% of its surface area is converted (orange); or, the loss can be only contained in the 100m buffer zone because less than 50% of its surface area is converted (yellow).*

**Figure 3.3:** Representation of direct and indirect ES loss due to a threshold behaviour assumption: the wet grasslands example.

bility, and noise disturbance that can cause the loss of the service over an extended area (Figure 3.2). The impact on a recreational area will thus depend on its distance from the infrastructure.

Based on these considerations, the annual value (V) of ES loss (ESL) for each route option of the infrastructure is as follows:

$$V(ESL) = \sum_{i=1}^{I}\sum_{k=1}^{K} D(LC_{ki}) \times (v(ES_{ki}) \times p(ES_{ki})) + \sum_{i=1}^{I}\sum_{k=1}^{K} I(LC_{ki}) \times (v(ES_{ki}) \times p(ES_{ki}))$$

In this equation, $D(LC_{ki})$ is the area considered as directly impacted for each land cover type $i$ ($i = 1, ..., 17$) and ES type $k$ ($k = 1, ..., 15$), depending on the land take we consider. Land take is about 100m around the infrastructure axis, the surface being converted into a fenced surface and it is consequently lost as a natural habitat for plants and animals (Seiler, 2002). If all the associated features, such as verges, embankments, slope cuttings, parking places, and service stations etc. are included, the total area designated for transport is likely to be several times larger than the land-take surface. $v(ES_{ki})$ is the annual value per biophysical unit for ES type $k$ supplied by land cover type $i$, and $p(ES_{ki})$ is the biophysical value per unit area for ES type $k$ generated by land cover type $i$. $I(LC_{ki})$ represents the impact zone not comprising the zones directly impacted by in a linear manner, that is indirectly. This parameter is declined in three ways.

First, $I(LC_{ki})$ can be the impact on ecosystem functions which depend on a minimum area for ES supply (threshold effect). The area considered as impacted is either (a) the entire ecosystem area for any impact on sensitive ecosystems (*e.g.* inland marshes, water bodies and peatbogs) and for some ES, and for other ecosystems if 50% or more of the ecosystem area is impacted (direct loss otherwise) as in Figure 3.3. Second, $I(LC_{ki})$ integrates the impact on landscape connectivity, *i.e.* an area considered as being lost for ES provision in the area (outside of the area of direct impacts) which suffers a loss of landscape connectivity (see Figure 3.4 in the case of hunting recreation). Finally, $I(LC_{ki})$ represents an impact on a point of interest, *e.g.* recreational sites, the area of loss considered is contained in the area corresponding to the area of exposure to noise and visibility.

### 3.2.3 Biophysical measurement of ecosystem service loss and economic values associated

Data were collected in the following ways. The biophysical and economic values per biome were selected where possible from the study site area. If no data were available, we applied benefit transfer from other French case studies or from European countries with similar climate, vegetation and socioeconomic characteristics. We selected economic values per biome based on an accepted economic valuation method: market prices, replacement costs, avoided damage costs. Finally, we used present values standardized in euros for the year 2010 (we employed a deflation with the general index of consumer prices published by public statistics (INSEE)). For each service, we describe the method to value the loss that is the difference in biophysical units and the economic loss associated between the baseline


**Figure 3.4:** Representation of species habitats and migration corridors threatened by the infrastructure construction

condition and the condition with project.

***Provision of picking products***

For wild foods (berries, flowers, mushrooms, chestnuts) produced by forests, we chose to retain the reference value given by Chevassus-au-Louis et al (2009), that is 15 €/ha/year. This value has been retained after the survey conducted by the French ministry in 2002 to evaluate the amount picking food for auto consumption (MAP/IFN, 2006). They found that for the year 2002 12.6 thousand tons of mushroom for auto consumption were collected, 4.4 thousand tons of fruits, and 330 tons of flowers. For hedgerow berry production is estimated approximately at 1kg/km/year and multiplied by their average berries market price (10 €/kg). We estimate the service at approximately 10 €/km/year.

***Raw material provisioning***

For forests, we assessed the average production per hectare of firewood by using the average firewood consumption of households in France (19 M$m^3$/year, INSEE survey) and the surface covered by forests in France (16 Mha, IGN, 2012) giving an average consumption of 1.16 $m^3$/ha. For hedges, we relied on surveyed data from hedge shredding sites (4 to 7

$m^3$/km/year; AILE, 2009). We monetized firewood according to the estimation made by Montagné and Niedzwiedz (2009) for the value of non-marketed firewood at 10.3 €/m3.

For fodder provisioning (benefiting to farmers), we relied on the average annual production of dry matter per hectare supplied by unseeded grasslands (1.5t/ha) and wetlands (4 t/ha) in France (AGRESTE-agricultural statistics, 2010; CGDD, 2011). To monetize the service we used the average fodder market price (representing the fodder savings), ranging from 55 to 110 €/ton of dry matter (CGDD, 2011). This value is equivalent to the price for putting at disposal a non-fertilized grassland (100 €/ha/year). For wetlands, we chose to evaluate the service with the highest market price, 110 €/ ton of dry matter because values founded in the literature are much higher (306 €/ha/year according to de Groot et al, 2012).

***Air quality regulation***

Urban and peri-urban forests contribute to air purification by filtering or eliminating a number of pollutants and particles. They consequently contribute to the environmental quality maintenance and health for urban and peri-urban citizens (McPherson et al, 1997). The gas removal is primarily done by absorption via leaf stomata, though some gases are removed by the plants' surface (Lovett et al, 1994). Trees also allow the partial interception of suspended particles, absorbed into the tree or retained on the plant surface, and then either re-suspended in the atmosphere, washed off by rain, or released into the ground with leaf fall (Nowak et al, 2006). The principal pollutants are: CO (carbon monoxide), nitrogen dioxide ($NO_2$), $O_3$ (ozone), particulate matter less than $10\mu m$ (PM10) and sulfur dioxide ($SO_2$). We relied on the Lovett (1994) model based on deposition velocity and local pollutant concentration to determine the amount of pollutant removed by peri-urban forests in on study area. To monetize the service we transferred the damage cost value for a marginal pollution change per ton of pollutant (for pollutants for which the threshold is beyond the regulatory concentration limit) in order to value the cost in terms of impacts on health, crops and materials. (British Department for Environment, Food and Rural Affairs (DEFRA) (Watkiss et al, 2006). Values are situated at 1.1 €/kg for $NO_2$, 55 €/kg for PM10 and 1.9 €/kg for $SO_2$ (Watkiss et al, 2006). For more information see Appendix B.1.

***Local climate regulation***

The local climate regulation service is supplied through two phenomena. The first is the cooling effect supplied by urban and peri-urban forests by reducing solar radiation, enhancing evapotranspiration and providing shade. Beneficiaries are then urban and peri-urban citizens. We assessed this service by using the avoided air conditioning cost (the reduced annual energy demand) due to the presence of urban forest that reduces energy demand for cooling from 2 to 7%/ year (McPherson et al, 1997). Air conditioning costs in France are about 14 €/households/ year (INSEE, national statistics), we estimate the avoided cost at about 0.63 €/household/year, and the value is calculated for households bordering urban forests.

Hedges also supply a local climate regulation service when they are situated on the edge of a culture by providing a windbreak. Here, the net effect of shelter on crop yield is positive, and the sheltered area (horizontal distance perpendicular to the hedge) is roughly proportional to hedge height (Vigiak et al, 2003). It is assumed that the protection can occur in an area 15 to 20 times the hedge height for an average height of about 3 meters and the productivity of this surface can be increased by between 5% and 30% compared to a situation without hedges (Brandle et al, 2004; Kort, 1988). We chose to monetize the service by approximating the production gain value of the crop adjacent to the hedge. We considered crop yield per hectare cultivated in the study area according to the AGRESTE-agricultural statistics (2010) and multiplied it by their prices in the market (excluding subventions and intermediate inputs values). We then produced an average value for protection of annual crops (ranging from [11-66] €/ha/year for zone 1 and to [13; 78] €/ha/year for zone 2), because we lacked data on the location of particular crops at a given location. For more information, see Appendix B.2.

***Global climate regulation***

The service is assessed as the (annual) carbon storage service loss due to soil sealing and the carbon stock released because of construction (what would have been gained if the stock had remained in the soil during the period instead of being released). The assessment was made as described in the Chapter 2. The biophysical loss was calculated by assuming that the actual stock is in equilibrium in order to avoid any carbon sequestration overestimate. The data of soil organic carbon and above ground and below ground vegetation carbon stock per ecosystem type were summed to assess the overall carbon stock released. We monetize the stock by using the opportunity cost of meeting mitigation policy goals in terms of costs and efficiency (Chevassus-au-Louis et al, 2009; Quinet et al, 2009).

***Flood protection***

Forests intercept periodical heavy rainfall, thus preventing rivers from flooding. This service is measured through the maximal potential interception during the largest rainfall event in the year (Biao et al, 2010). The maximum potential interception (in $m^3$) can be estimated by summing the rate of canopy interception (C), litter (L) and soil (S) retention, where C depends primarily on forest type and the amount of the largest rainfall event, L depends on the forest type as well as the thickness of the litter layer, and S depends principally on the forest type and soil depth. The service is valued by using replacement costs that is the annual amortization cost of a reservoir. According to Guinaudeau (2009), if we apply an amortization cost of 4% over 25 years, the cost returns are (0.3- 0.4) €/$m^3$/year.

Wetlands situated in the floodplain of a river enable excess water to spread out over a wide area during a flood event, which reduces the speed and volume of runoff, thus limiting or preventing flood damage. Here, the economic valuation relies on benefit transfer from French avoided flooding damages costs studies (Agence de l'eau Adour-Garonne, 2009; Cachard-berger, 2000; Laurans et al, 1996; Laurans and Argaud, 1999). For more information, see Appendix B.3.

***Erosion prevention***

The erosion prevention service estimates the ability of a landscape and particularly of vegetation to retain soil. The service is typically calculated as a function of vegetation cover, topography and soil erodibility, integrated in the Universal Soil Loss Equation (USLE), which is the most often used (Crossman et al, 2013; Wishmeier and Smith, 1978). We used the InVEST software model (Kareiva et al, 2011), ran with and without the infrastructure to calculate avoided erosion due to vegetation retention between both land uses. The service can be seen as a maintenance of arable land, and to value it we used the replacement cost value given by Leonard (2009) which varies between 10 and 15 €/ ton. For more information, see Appendix B.4.

### *Pollination*

Most fruits, vegetables and oilseed crops are dependent on pollination services performed by pollinators (Klein et al, 2007). Using the InVEST model for crop pollination, we calculated a score of pollinator abundance for each cell according to the availability of nest sites and floral resources supplied by the landscape and the average distance travelled by different pollinators' species (Kareiva et al, 2011). Flower resources in nearby cells are given more weight than distant cells, according to the species' average foraging range and an exponential decrease with distance. To take into account the demand that is agricultural covers needing pollination, the abundance score is summed for cells surrounding agricultural covers giving more weight to nearby cells (decreasing exponentially with distance). Data were collected from literature review (Londsorf et al, 2009) and readapted to the area according to expert opinion. The result is finally a score ranging from 0 to 1 combining pollinators "supply" (abundance) and "demand" for the agricultural covers. We consider that the contribution of pollinators to crop yield is about 10% (Gallai et al, 2009), then to monetize the service we multiply the pollination score (from 0-1), to 10% of the average crop production value in the area (AGRESTE agricultural statistics (2010), excluding subventions and intermediate inputs values). The values of crop productions calculated for the local climate regulation are then used again here, and are about 109 €ha for orchards, and about 24€/ha/year, and 27€/ha/year for annual crops respectively for Zone 1 and Zone 2. For comparison the values proposed by the Chevassus-au-Louis (2009) report were ranging from [60-80]€ /ha/year. We ran the model with and without the infrastructure to assess the loss associated with its construction. For more information, see Appendix B.5.

### *Biological control*

Natural control of plant pests is supplied by many different species, including birds, bats, spiders, beetles, mantises and flies (TEEB, 2010). This service involves two types of benefits: in the short-term, it suppresses pest damage and improves crop yields, and in the long-term it maintains an ecological equilibrium that prevents herbivorous insects from reaching pest status (TEEB, 2010). Data on populations of biological control agents are limited but the trend of this service is presumed to be negative owing to habitat conversion. Even though the relationship between densities of natural enemies and the biological control services they provide is not likely to be linear, we relied on benefit transfer given the lack of data. Values for grasslands and fruit trees, vineyards, and olive groves were derived from the study of Brenner-Guillermo et al (2007) for Spain that is 24 €/ha/year. For forests, the economic value of the service is derived from replacement costs in Sweden (Hougner et al, 2006) and avoided damage costs in China (Xue and Tisdell, 2001) to estimate the service at 169 €/ha/year for the year 2010. Finally, the value for wetlands was based on the study of Everard and Jevons (2010) in the UK, who assumed that biological control accounts for a part of crop yields and valued the service at 134 €/ha/year.

***Recreation***

We dissociated fishing and hunting recreation from more general outdoor recreation, because both services are considered to be related to suitable living space (habitats and migration) for animals and thus require a different form of analysis.

*General outdoor recreation:* Outdoor recreation is evaluated principally with benefit transfer weighted by the location of the natural or semi-natural site: proximity to a city, city size, abundance of similar sites in the study area, or recognized, or at less than 100m from the site, as a touristic area (or containing cycling routes, green routes, or horse riding routes, outdoor recreation equipment). For grassland, we used the value proposed in de Groot et al (2012), that is 19 €/ha/year. We applied the value only to grassland identified as a touristic area and situated at less than 2 km from an urban area.

For forests, we relied on the reference value proposed by Chevassus-au-Louis et al (2009) of 200 €/ha/an, which corresponds to the total expenses incurred in terms of travel costs (for an average of 18 trips per year at roughly 4.5 €per trip) with respect to the entire French forest area (about 58 trips/ha/year). A multiplying factor is applied, as suggested by the same report, according to the forest accessibility, proximity from urban areas and abundance in the landscape. These different criteria will impact the number of trips. The multiplying factors are defined as follow:

- 0 private forests;
- 1 public forests > 20 km from an agglomeration (< 20,000 inhabitants);
- 2 public forests < 20 km from an agglomeration (< 20,000 inhabitants);
- 3 public forests < 20 km from an agglomeration (>200,000 inhabitants);
- 4 peri-urban forests < 20 km from an agglomeration (> 200,000 inhabitants with other forest cover in the sector);
- 5 peri-urban forests < 20 km from an agglomeration (> 200 000 inhabitants and if it is the only forest in the area).

The estimated value varies from 0 to 1,000 €/ha/year. For comparison, the value given by Groot et al (2012) is about 698 €/ha/year.

Wetlands were selected when they were specifically identified as a touristic site. We applied a mean value to the selected wetlands from the Brander et al (2006) meta-analysis based on 89 studies, that is 469 €/ha/year. The value assigned to the wetland recreation is also attributed to water bodies as proposed by Brander et al (2006) for inland wetlands. Finally for the recreation value of agricultural areas, we relied on the mean value 29.8 €/ha/year based on two contingent valuations for UK and USA (Alvarez-Farizo et al, 1999; Bergström et al, 1985).

*Fishing recreation:* For fishing, we used national statistics (AGRESTE-Water survey) which estimated this ES at between 170 and 337 €/ha/year. For water bodies, we calculated the value of this ES based on the number fishing permits sold in the study area in the year 2011, and the surface occupied by watercourses and water bodies. We obtained a value of 125 €/ha/year for the eastern part of the study area, and 186 €/ha/year for the western part. For comparison the values used by other authors range from is 76 €/ha/year (de Groot 2012) to 353 €/ha/year (Brander et al, 2006) and from [80; 120] €/ha/year in 15 French Studies reported by CGDD (2010).

*Hunting recreation:* To model the loss of this service we used the OptiFlux software for large mammals, a spatial analysis tool designed to predict and visualize the effects of implementing a linear infrastructure upon wildlife habitats (see Baghli and Thiévent, 2011). Optiflux uses data on ecological requirement of particular animal species (preferential habitats, role in the species ecology, feeding, breeding, migration, etc.) and resistance coefficient of the landscape. The calculation is made before and after the infrastructure construction to assess the loss of habitat and its effect on movement for game provisioning. To evaluate the benefit loss, *i.e.* the loss of territory for hunting permits, we used the reference value of 62 €/ha/year proposed by Chevassus-au-Louis et al (2009). This value represents the total expenses realized by hunters over the total surface area where hunting permits are delivered in France. For more information on the Optiflux principles, see Appendix B.6.

## 3.3 Results

Our study considers different alternative routes in two zones crossed by a high-speed rail project in France. The route alternatives were chosen because they are those for which a choice had been made during the environmental studies and for which discriminating criteria to make the choice were lacking. The first zone has three alternative routes and the second zone two alternative routes. The first zone is characterized by agricultural plains and a medium sized urban area surrounded by a forested belt. The second zone is characterized by a more important relief, fields separated by hedges, plateaus, and several small inland wetlands (Figure 3.5). The longest route is route 1.3 for the first zone, and route 2.2 for the second zone. The route options produce different ES losses (Table 3.2).

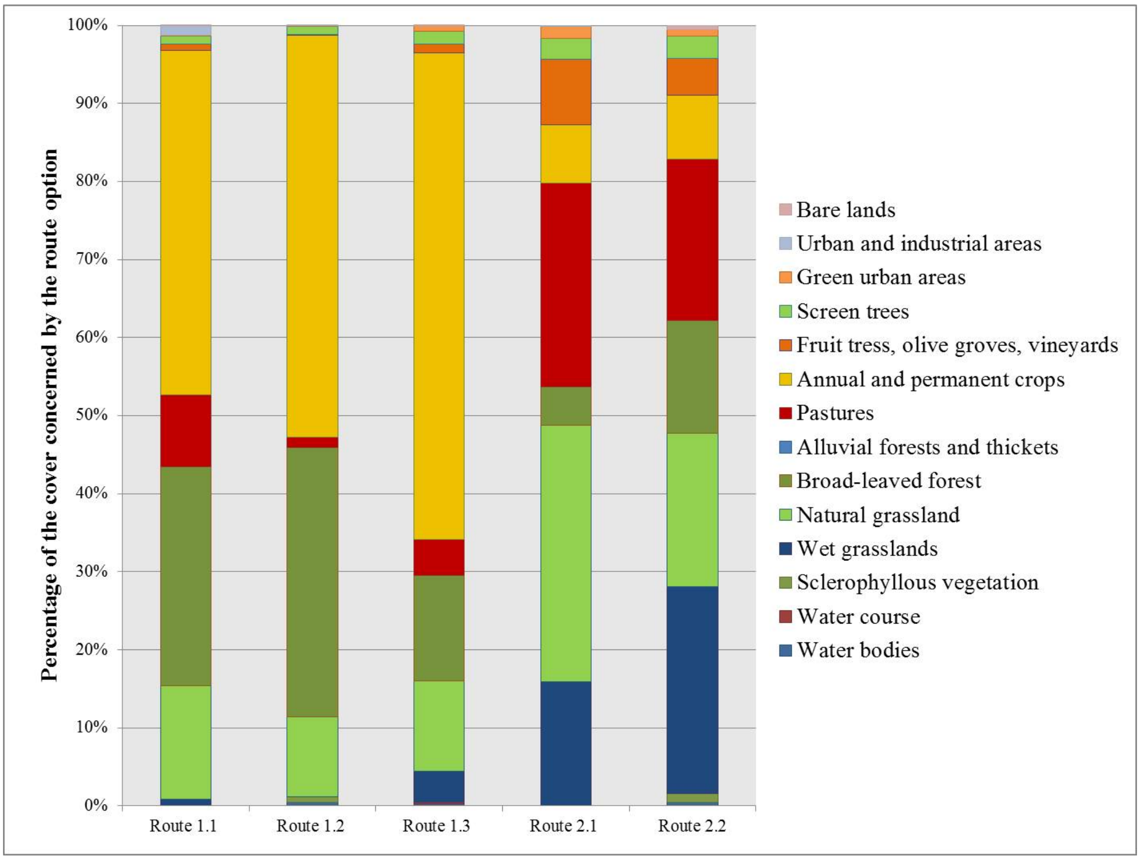



**Figure 3.5:** Share of ecosystem impacted by different route options (with a 100m land take assumption)

For zone 1, the least impacting route is route 1.3 (the slightly longest route) followed by route 1.1, and the most impacting route is route 1.2. This result remains the same when the global climate regulation service (the most valued service) is not assessed and when

**Table 3.2:** Annual economic loss per service and per route option

| | Zone 1 | | | Zone 2 | |
|---|---|---|---|---|---|
| In euros for the year 2010 | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
| Picking products | 1, 119 | 1, 295 | 806 | 1, 262 | 1, 746 |
| Raw materials | 1, 953 | 1, 662 | 2, 348 | 6, 050 | 8, 244 |
| Total | 3, 072 | 2, 958 | 3, 154 | 7, 312 | 9, 990 |
| | | | | | |
| Air quality | 90, 014 | 112, 092 | 36, 362 | - | - |
| Local climate | 1, 426 | 1, 188 | 1, 164 | 1, 778 | 1, 737 |
| Global climate | 107, 936 | 109, 862 | 106, 956 | 137, 671 | 187, 611 |
| Flood protection * | 1, 577 | 1, 902 | 1, 310 | 870 | 1, 203 |
| Water flow regulation * | - | - | 604 | 1, 588 | 2, 544 |
| Erosion prevention | 17, 600 | 8, 750 | 4, 450 | 16, 775 | 16, 375 |
| Pollination * | 3, 658 | 3, 363 | 3, 354 | 28, 359 | 5, 188 |
| Biological control | 10, 182 | 11, 934 | 6, 837 | 7, 986 | 14, 832 |
| Total | 232, 394 | 249, 092 | 161, 037 | 195, 026 | 229, 490 |
| | | | | | |
| Recreation * | 10, 194 | 10, 271 | 19, 625 | 18, 068 | 19, 060 |
| Fishing recreation * | 16 | 95 | 957 | 4 457 | 7 004 |
| Hunting recreation * | 36, 227 | 30, 820 | 43, 481 | 20, 929 | 16, 530 |
| Total | 46, 437 | 41, 186 | 64, 064 | 43, 454 | 42, 594 |
| | | | | | |
| Route Length (km) | 20.19 | 20.01 | 22.34 | 21.59 | 22.62 |
| Annual loss per route alternative | 281, 903 | 293, 235 | 228, 254 | 245, 793 | 282, 073 |
| Loss per km | 13, 957 | 14, 651 | 10, 217 | 11, 935 | 12, 465 |

* Service with an indirect supplementary loss considered
*Shaded cells highlight the least impacting routes*

we assess the result per kilometer. This is an important result because it means that the longest route is not automatically the one that produces the most significant loss in ES. For provisioning services, route 1.2 has a slightly lower impact than the others, primarily because this is the route which least affects natural grasslands that provide fodder. The analysis gives however a lower importance to these services compared to the regulation and cultural services. For regulation services, route 1.3 is the least impacting route for all ecosystem services, except for water flow regulation because it is the only route option that crosses small wetlands related to a water system. This result was to be expected since route 1.3 is the one that contains the least natural and semi-natural ecosystems. Air quality is responsible for a large part of the total value, a result that can be explained by the fact that the first two routes cross an important amount of peri-urban forests (which are important elements of this ES). However, regarding cultural services, route 1.3 involves the highest loss, due to its high impact on landscape connectivity. Overall, for zone 1, route 1.3 represents the best choice for the maintenance of the majority of ES, leading to an annual loss of approximately 228,254 Euros (in Euro 2010), while route 2 involves a loss of 293,235 Euros.

For zone 2, the least impacting route is route 2.1, which causes an annual loss of 245,793 euros (route 2.2 involves a loss of 282,073 euros 2010). This result is also true for the loss per kilometer. This route alternative contains less natural and semi natural ecosystems (grasslands, forests and wetlands) than the other route options. This trade-off only changes for erosion prevention, pollination and for hunting recreation services. This can be explained by the fact that route 2.1 crosses more croplands that require pollination services (particularly fruits trees), has more nesting habitats in its proximity and has a greater effect on landscape connectivity than route 2 (a negative effect that can be reduced by the construction of wildlife passageways). Overall, route 2.1 has the least impact in zone 2, however it has a greater impact on landscape connectivity, raising the question of the relative importance of different ecosystem services.

We also tested different ways of assessing effects that impinge on the alternatives ranking of different route options. First, the effects of a change in LULC data was tested by assessing the loss with a less precise typology, *i.e.* Corine Land Cover (at a scale of 1:25000). Using these data we observed a very similar ranking of route options in relation to ES loss, but also conduce a global underestimation of the loss, ranging from 2-20% less loss than with the more precise land cover data (Figure 3.6). The ES underestimated with the Corine Land Cover classification (picking products, raw materials, local climate, water flows, fishing recreation) concern very small areas that were not detected with this less precise classification. Conversely, an overestimation is made by estimating the erosion prevention service with Corine Land Cover. This can be explained by the fact that Corine Land Cover considers larger areas supplying the ES than the Corine Biotope classification, and the service is thus overestimated (This result has been also observed in the study conducted by Kandziora et al, 2013). Then we can see that using a more precise typology is particularly important in estimating raw materials service, local climate regulation service, water flows, fishing recreation, or pollination service.

For the sensitivity to mean values, we tested the effect of assessing the ES loss with the lower value in terms of combination of the lower biophysical value and economic value, and the higher value we observed (see Table 3.3). This analysis produced the same trade-off for all type of services with route 1.3 (zone 1) and route 2.1 (zone 2) being the least impacting routes for the provision and regulation ES, and the most impacting route for recreation. The analysis of the overall annual loss (for every services) and the loss per kilometre gives the same ranking for lower and higher values for estimates.

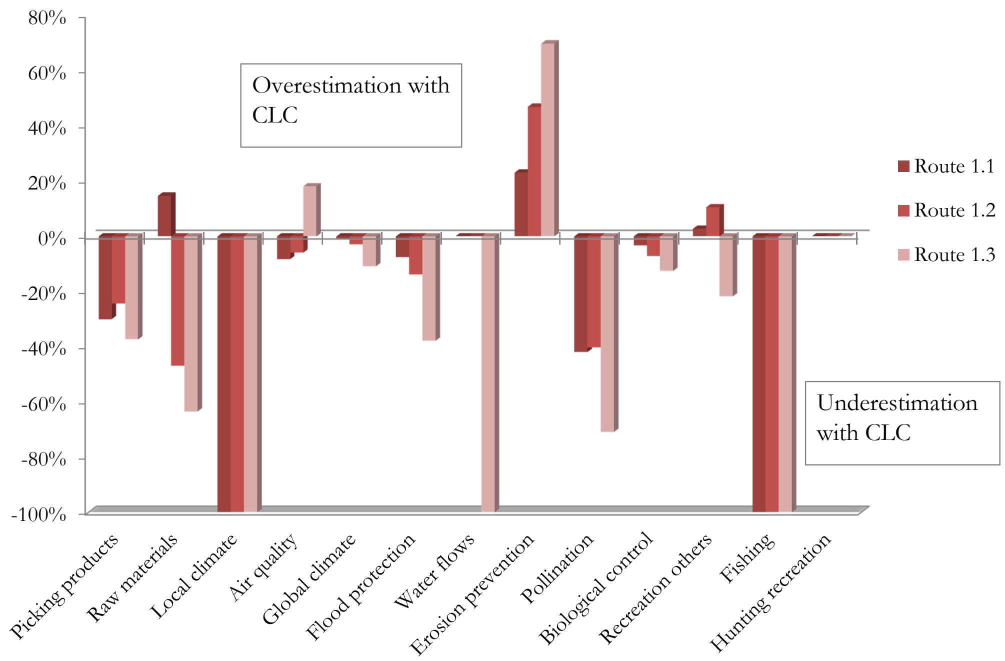



**Figure 3.6:** Differences between an estimation of ES loss with the Corine Biotope Classification (scale 1:5000) and with Corine Land Cover data (scale 1:25000) for the first zone

**Table 3.3:** Ranking sensitivity to mean values: annual loss when estimates are made with the lower combination and higher combination of biophysical and economic values

| | Zone 1 | | | | | |
|---|---|---|---|---|---|---|
| | Route 1.1 | | Route 1.2 | | Route 1.3 | |
| | Low value | High value | Low value | High value | Low value | High value |
| Annual loss (in euros 2010) | 225, 276 | 453, 775 | 235, 323 | 480, 302 | 173, 858 | 423, 923 |
| Annual loss/km | 11, 152 | 22, 464 | 11, 760 | 24, 003 | 7, 782 | 18, 976 |
| | Zone 2 | | | | | |
| | Route 2.1 | | Route 2.2 | | | |
| | Low value | High value | Low value | High value | | |
| Annual loss (in euros 2010) | 229, 807 | 471, 680 | 248, 145 | 525, 971 | | |
| Annual loss/km | 10, 644 | 21, 847 | 10, 965 | 23, 242 | | |
| *Shaded cells highlight the least impacting routes* | | | | | | |

## 3.4 Discussion

In this study we provide an examination of how to quantify both direct and indirect impacts on ES provision for different options associated with the implantation of a linear infrastructure. Except for the flood protection service, the ES analysed here are not currently integrated into environmental impact assessment. Our study thus provides an initial attempt to integrate such services. We do so in a way which points out the necessity of discriminating impacts which may have a direct effect on ecological function (and thus ES) or an indirect effect because of impacts on landscape connectivity if the infrastructure creates a barrier or impacts habitats that have ecological interactions with other landscape elements. Finally, we provide a preliminary illustration of how to integrate the fact that both direct and indirect impacts may be associated with a form of threshold behaviour in ecosystem function. The results of our study are discussed in terms of the relevance of such information for EIA, consultation with the general public and informing stakeholders.

### 3.4.1 Landscape connectivity

Few studies have developed conceptual or theoretical frameworks to link landscape connectivity with the provision of ecosystem services (Mitchell et al, 2013). There are two important points here. First, it is necessary to identify what types of connectivity might affect ecosystem service provision. Second, the possible mechanisms by which connectivity might affect ecosystem service provision, either directly and indirectly have rarely been explicitly identified or measured. This distinction is important because loss of connectivity can affect ecosystem service provision directly by impeding movement of organisms and matter through a landscape, but also indirectly by altering levels of biodiversity and ecosystem function in different but interdependent areas.

In the examination of the different route options, it appears that the area of loss considered can be critical in the analysis. By taking into account the indirect effects of infrastructure construction on ES supply, the ranking of alternative routes in terms of direct losses alone is changed for some services (particularly for flood protection and hunting recreation). Indeed, the route option (2.1) that incurs the lowest loss of ES in relation to direct surface losses in zone 2 is the option that incurs the greatest loss on landscape connectivity. For the flood protection service, the tradeoff consistently changes when only direct impacts are considered such as the least impacting route (route 1.3) become the more impacting route when we consider only direct impacts. Overall, such changes may cause between 10 and 80% of differences depending on the ES. This highlights the importance of identifying the extent to which indirect impacts can cause ES loss or the importance of landscape connectivity. A result that may be even more critical if particular habitats that are impacted play a key role in movement patterns (Gaaff and Reinhard, 2012), particularly when movements of large mammal (Forman and Deblinger, 1999; Hilty et al, 2006) or species that incur high dispersal mortality (Flather and Bevers 2002) are affected.

As Kettunen and ten Brink (2006, page 19) wrote:

> "Habitat alteration and destruction appear to be the most common direct reasons behind the loss of biodiversity and related ecosystem services. Additionally, over-extraction of resources, pollution and eutrophication, and changes in ecosystem species composition (introduction of invasive alien species) have often contributed to the loss."

Further, ten Brink et al (2008) argue that below a certain level, areas of habitat will not sustain certain species, with a consequent loss in terms of game availability, diversity and migration paths. This is related to the fact that ES that depend on landscape connectivity, which can be greatly affected by fragmentation due to linear infrastructures (see section 2.3). Terrestrial transport infrastructure construction can involve all of the drivers of loss cited here. Hence, the consideration of indirect impacts on the ecosystem functions that maintain ES supply at a "desired level" is an important but difficult question, which has to be studied in an interdisciplinary perspective (Groffman et al, 2006).

### 3.4.2 Threshold behaviour

By testing the effects of incorporating a scenario in which small scale ecosystems show a rapid threshold behaviour in terms of the loss of ES due to direct and indirect impacts we illustrate, albeit in a preliminary fashion, a means of adopting a precautionary approach, determined by the risk of loss. It is a preliminary and precautionary approach because the true level of the threshold and the associated external cost are not precisely known (Perrings and Pearce, 1994; Huggett 2005; Lindenmayer and Luck 2005).

Many simulations have suggested that increasing fragmentation effects at low levels of habitat can produce threshold relationships with habitat proportion. King et al (2005) show threshold relationships between watershed land cover and the condition of stream ecosystems. Partial correlation analysis of land-cover percentages revealed that simple correlations described relationships that could not be separated from the effects of other land-cover classes or relationships that changed substantially when the influences of other land-cover classes were taken into account. Further analyses revealed that spatial arrangement of land cover, as measured by areal buffers and distance weighting, influenced the amount of developed land, resulting in a threshold change in macroinvertebrate-assemblage composition. Sudding and Hobbs (2009) illustrate that an essential part of the decision-making process is evaluating the evidence for, and the uncertainty of, threshold behaviour in a given management situation. Although it might be impossible to rigorously test many of the assumptions of threshold models, we suggest the need for an increased emphasis on their potential occurrence. Reviews of this issue (Swift and Hannon 2010) indicate that threshold model are highly applicable in managed systems and that human-impacted habitats can be particularly susceptible to threshold shifts. However, it is likely that not all systems exhibit threshold dynamics and that there are both costs and benefits to their incorporation in management frameworks. Despite the difficulty of rigorous testing of theoretical assumptions of threshold models in applied settings, theory is beginning to provide tools for the evaluation of evidence and the uncertainty of threshold behaviour in a given management situation (Sudding and Hobbs 2009). The study of critical thresholds in landscape ecology raises many questions which remain for future research in relation to their generality and how commonly they occur, their causes, and their precise impacts (Huggett 2005, Lindenmayer and Luck 2005).

### 3.4.3 Stakeholder consultation

Examined in the light of results used for the choice of route option in the project for the installation of the infrastructure under question, ours results illustrate a certain degree of coherence with stakeholder choices. For zone 1, and at the beginning of the process, the majority of the stakeholders retained the route option 1.2 based on technical characteristics. However, the EIA showed that route 1.3 was the most favourable with regards to all environmental issues. This is in accordance with our results in terms of ES loss based on the assumptions used for the analysis.

For zone 2, the first alternative route involved passing near a relatively large provincial town, thus engendering noise effects and other nuisances, whilst route 2.2 engendered more environmental effects, a result confirmed by the assessment of ES loss. However, our results also showed that route 2.2, in addition to a significant environmental impact, involves a more important loss of the recreation services currently supplied to urban areas by certain ecosystems, an impact which is not integrated in the absence of an assessment of ES loss. Hence, in this case study and for both zones, the ES loss criteria could provide a novel and complementary information for assessing environmental impact and decision making.

### 3.4.4 Limits to the quantification of ES

In the absence of site-specific valuation data we relied, for some ES, on the benefit transfer method. This method uses economic information collected for a given area (study site) at a given time to make inferences on environmental goods and services in a different location (application site or policy site) and at a different time but for the same ecosystem type. Some authors criticize this approach because it relies on a number of assumptions concerning the equivalence among ES supply and value at different sites. For example, Eigenbrod (2010) suggests that the errors associated with ES mapping based on the benefit transfer method are likely to be high, primarily because of generalization errors. These errors can be attributed to the extrapolation of economic values between sites that may be very different in terms of their social, demographic or economic contexts. Indeed, different sites can have different markets; hence prices and substitute price relationships will vary (Loomis and Rosenberg, 2006). In addition, inconsistency of biophysical measures may generate large errors when translating across sites (Bateman et al, 2006). We thus recognize that the results based on a benefit transfer method should be interpreted with caution, and we have minimized as much as possible the generalization error by adapting values to the local context by using as much local information as possible. In this respect, we used areas to provide data that mostly involve French or other European studies or countries classified as high income countries as part of the OECD, and with high a density of population by the World Bank.

## 3.5 Conclusions

Our study identifies four critical steps for the integration of ES loss into EIA of linear infrastructure projects: (i) the identification of potential ES supplied and impacted in a given landscape, (ii) the identification of the area of ES loss and in particular its direct or indirect character, (iii) the biophysical assessment and economic valuation of the loss, and (iv) the mapping and calculation of ES loss for different route options in order to compare them. The conversion of natural habitats and ES loss are currently poorly assessed and valued in monetary terms in EIAs for infrastructure construction.

In our study we produced a classification in which ES loss may result from both direct and indirect (landscape connectivity) impacts on ecosystem function. .In addition, because ecological thresholds may be crossed for ES supply when even a small part of an ecosystem is converted into an artificial surface we provide a precautionary, and preliminary, integration of the possibility of such threshold behaviour in relation to the scale of ecosystem functions. In any assessment of ES supply and loss it is becoming clear, despite the context-dependence of such behaviour, that some consideration of such threshold behaviour in ecosystem function should be integrated into the assessment of ES loss. In addition to identifying the key aspects of ecosystem structure and function that influence ES provision, and how impacts may directly or indirectly impact ecosystem function and ES provision, identifying whether the factors and their interactions exhibit threshold responses is also important.

Our study thus provides an original attempt to broaden the scope of terrestrial transport infrastructure project evaluation by incorporating ES loss into a more global consideration of environmental losses. The different options assessed in our study provide an example of how decision-making regarding route selection can be improved by mapping ES loss. This type of analysis may provide a more precise assessment of the socio-economic implications of the environmental impact of infrastructure projects on a landscape scale, allowing for more efficient control of natural capital loss. This may also allow decision-makers to confront costs linked to ES loss to the more traditionally quantified benefits of such project in terms of time gains. This kind of analysis could be used more generally to assess linear infrastructure projects simply by adjusting the framework in relation to the types of project and ecosystems that are concerned.

## References


Agence de l'eau Adour-Garonne, 2009. Evaluation économique des zones humides, Etudes De cas sur le bassin Adour Garonne. Ecowhat-Actéon, France.

AGRESTE, 2010. Memento de la production agricole, online, avalaible in : `http://agreste.agriculture.gouv.fr/`.

AGRESTE- SOeS - SSP, Enquêtes eau et assainissement 2008, 2004 et 2001 et Insee - Recensement de la population. France.

Alvarez-Farizo, B., Hanley, N., Wright, R.E., MacMillan, D., 1999. Estimating the benefits of agri-environmental policy : econometric issues in open-ended contingent valuation studies. Journal of Environmental Planning and Management 42, 23-43.

Andrén, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat : a review. Oikos 71, 355-366.

Arhington, A.H., Naiman, R.J., McClain, M.E., Nilsson, C., 2009. Preserving the biodiversity and ecological services of rivers : new challenges and research opportunities. Freshwater Biology.

Baghli, A., Thiévent, P., 2011. OptiFlux : when biological and spatial data are crossed to reduce effects of linear infrastructure on wildlife. Procedia Environmental Sciences 9, 11-15.

Bagstad, K.J., 2009. Ecological economic applications for urban and regional sustainability, Natural Resources. University of Vermont, Burlington.

Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows : A comprehensive approach to quantifying actual services. Ecosystem Services 4, 117-125.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecological Monographs 81, 169-193.

Bateman, I.J., Day, B.H., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values : Welfare measures, distance decay and total WTP. Ecological Economics 60, 450-460.

Beier, P., 1995. Dispersal of juvenile cougars in fragmented habitat. Journal of Wildlife Management 59, 228-237.

Bergstrom, J.C., Dillman, B.L., Stoll, J.R., 1985. Public environmental amenity benefits of private land : the case of prime agricultural land. South Journal of Agricultural Economics 7, 139-149.

Biao, Z., Wenhua, L., Gaodi, X., Yu, X., 2010. Water conservation of forest ecosystem in Beijing and its value. Ecological Economics 69, 1416-1426.

Brander, L.M., Florax, J.G.M., Vermaat, J.E., 2006. The empirics of wetland valuation : a comprehensive summary and meta-analysis of the literature. Environmental and Resource Economics 33, 223-250.

Brandle, J.R., Hodges, L., Zhou, X.H., 2004. Windbreaks in North American Agricultural Systems, Agronomy and Horticulture. Faculty Publications-University of Nebraska, Lincoln.

Brauman, K.A., Daily, G.C., Duarte, T.K., Mooney, H.A., 2007. The nature and value of ecosystem services : an overview highlighting hydrologic services. Annual Review of Environmental Resources 32, 67-98.

Brenner-Guillermo, J., 2007. Valuation of ecosystem services in the Catalan coastal zone, Marine Sciences. Polytechnic University of Catalonia, Barcelona.

Broekx, S., Liekens, I., Peelaerts, W., De Nocker, L., Landuyt, D., Staes, J., Meire, P., Schaafsma, M., Van Reeth, W., Van den Kerckhove, O., Cerulus, T., 2013. A web application to support the quantification and valuation of ecosystem services. Environmental Impact Assessment Review 40, 65-74.

Brosi, B.J., Armsworth, P.R., Daily, G.C., 2008. Optimal design of agricultural landscapes for pollination services. Conservation Letters 1, 27-36.

Brown, L. C. and Foster, G. R. : Storm erosivity using idealized intensity distributions, Transactions of the Asae, 30, 379-386, 1987.

Burkhard, B., Kroll, F., Müller, F., Windhorst, W., 2009. Landscapes' Capacities to Provide Ecosystem Services - a Concept for Land-Cover Based Assessments. Landscape Online 15, 1-22.

Cachard-Berger, I., 2000. Evaluation économique des services rendus par le marais de la souche, PNRZH, France.

Carré, G., Roche, P., Chifflet, R., Morison, N., Bommarco, R., Harrison-Cripps, J., Krewenka, K., Potts, S.G., Roberts, S.P.M., Rodet, G., Settele, J., Steffan-Dewenter, I., Szentgyörgyi, H., Tscheulin, T., Westphal, C., Woyciechowski, M., Vaissière, B.E., 2009. Landscape context and habitat type as drivers of bee diversity in European annual crops. Agriculture, Ecosystems and Environment 133, 40-47.

Chevassus-au-Louis, B., Salles, J.M., Pujol, J.L., Bielsa, S., Martin, G., Richard, D., 2009. Approche économique de la biodiversité et des services liés aux écosystèmes : contribution à la décision publique. La Documentation Française, Centre d'Analyse Stratégique, Paris. (Online) Available from : `http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/094000203/0000.pdfS.`

Commissariat Général au développement durable, 2011. Evaluation économique des services rendus par les zones humides - Enseignements méthodologiques de monétarisation, Etudes et documents, Paris.

Commissariat Général au Développement Durable (CGDD), 2010. Projet de caractérisation des fonctions écologiques des milieux en France. Etudes et Documents 20-2010. Paris.

Daly, H., Farley, J., 2003. Ecological Economics : Principles and applications. Island Press, Washington, DC.

de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., Hussain, S., Kumar, P., McVittie, A., Portela, R., Rodriguez, L.C., ten Brink, P., van Beukering, P., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services 1, 50-61.

Eade, J.D.O., Moran, D., 1996. Spatial Economic Valuation : Benefits Transfer using Geographical Information Systems. Journal of Environmental Management 48, 97-110.

Eigenbrod, F., Armsworth, P.R., Anderson, B.J., Heinemeyer, A., Gillings, S., Roy, D.B., Thomas, C.D., Gaston, K.J., 2010. The impact of proxy-based methods on mapping the distribution of ecosystem services. Journal of Applied Ecology 47, 377-385.

Everard, M., Jevons, S., 2010. Ecosystem services assessment of buffer zone installation on the upper Bristol Avon. Environment Agency, Wiltshire.

Fahrig, L., 2002. Effect of habitat fragmentation on the extinction threshold : a synthesis. Ecological Applications 12, 346-353.

Fahrig, L., Rytwinski, T., 2009. Effects of roads on animal abundance : an empirical review and synthesis. Ecology and Society 14, 21.

Forman, R.T.T., 2000. Estimate of the Area Affected Ecologically by the Road System in the United States. Conservation Biology 14, 31-35.

Flather, C.H., Bevers, M. 2002. Patchy reaction-diffusion and population abundance : the relative importance of habitat amount and arrangement. The American Naturalist 159, 40-56.

Forman, R.T.T., Alexander, L.E., 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29, 207-231.

Gaaff, A., Reinhard, S., 2012. Incorporating the value of ecological networks into cost-benefit analysis to improve spatially explicit land-use planning. Ecological Economics 73, 66-74.

Gascoigne, W.R., Hoag, D., Koontz, L., Tangen, B.A., Shaffer, T.L., Gleason, R.A., 2011. Valuing ecosystem and economic services across land-use scenarios in the Prairie Pothole Region of the Dakotas, USA. Ecological Economics 70, 1715-1725.

Geneletti, D., 2006. Some common shortcomings in the treatment of impacts of linear infrastructures on natural habitat. Environmental Impact Assessment Review 26, 257-267.

Geneletti, D., 2011. Reasons and options for integrating ecosystem services in strategic environmental assessment of spatial planning. International Journal of Biodiversity Science, Ecosystem Services and Management 7, 143-149.

Geneletti, D., 2013. Assessing the impact of alternative land-use zoning policies on future ecosystem services. Environmental Impact Assessment Review 40, 25-35.

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., LeRoy Poff, N., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds : the key to successful environmental management or an important concept with no practical application ? Ecosystems 9, 1-13

Guinaudeau, M., 2009. Retenues de substitution : sortir de la controverse. Agriculture, Alimentation et Espaces Ruraux 30, 5-6.

Helming, K., Diehl, K., Geneletti, D., Wiggering, H., 2013. Mainstreaming ecosystem services in European policy impact assessment. Environmental Impact Assessment Review 40, 82-87.

Hicks, C., Peymen, J., 2002. Habitat Fragmentation due to Existing Transportation Infrastructure, in : Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J. (Eds.), COST 341 - Habitat Fragmentation due to transportation infrastructure : The European Review. Office for Official Publications of the European Communities, Luxembourg, pp. 73-113.

Hilty, J.A., Lidicker, W.Z.J., Merenlender, A.M., 2006. Corridor Ecology : The science and practice of linking landscapes for biodiversity conservation. Island Press, Washington, DC.

Holmlund, C.M., Hammer, M., 1999. Ecosystem services generated by fish populations. Ecological Economics 29, 253-268.

Honrado, J.P., Vieira, C., Soares, C., Monteiro, M.B., Marcos, B., Pereira, H.M., Partidário, M.R., 2013. Can we infer about ecosystem services from EIA and SEA practice ? A framework for analysis and examples from Portugal. Environmental Impact Assessment Review 40, 14-24.

Hougner, C., Colding, J., Söderqvist, T., 2006. Economic valuation of a seed dispersal service in the Stockholm National Urban Park Sweden. Ecological Economics 59, 364-374.

Huggett, A.J., 2005. The concept and utility of "ecological thresholds" in biodiversity conservation. Biological Conservation 124, 301-310.

IGN, 2012. La forêt en chiffres et en cartes, campagnes d'inventaire 2008-2011. Service de l'inventaire forestier statistique, France.

Kandziora, M., Burkhard, B., MÏler, F., 2013. Mapping provisioning ecosystem services at the local scale using data of varying spatial and temporal resolution. Ecosystem Services 4, 47-59.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E., 2011. Natural capital : Theory and practice of mapping ecosystem services. Oxford University Press, Oxford, UK.

Kettunen, M., ten Brink, P., 2006. Value of biodiversity- Documenting EU examples ; where biodiversity loss has led to the loss of ecosystem services, in : Institute for European Environmental Policy (IEEP) (Ed.), Final report for the European Commission, Brussels, Belgium.

King, R.S., Baker, M.E., Whigham, D.F., Weller, D.E., Jordan, T.E., Kazyak, P.F., Hurd, M.K., 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15, 137-153.

Klein, A.M., Vaissiere, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Tscharntke, T., 2007. Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B-Biological Sciences 274, 303-313.

Kort, J., 1988. 9. Benefits of windbreaks to field and forage crops. Agriculture, Ecosystems & Environment 22-23, 165-190.

Kouli, M., Soupios, P., Villianatos, F., 2009. Soil erosion prediction using the Revised Universal Soil Loss Equation (RUSLE) in a GIS framework, Chania, Northwestern Crete, Greece. Environmental Geology 57, 483-497.

Kremen, C., N. M. Williams, Bugg, R.L., Fay, J.P., Thorp, R.W., 2002. The area requirements of an ecosystem service : crop pollination by native bee communities in California. Ecology Letters 7, 1109-1119.

Kreuter, U.P., Harris, H.G., Matlock, M.D., Lacey, R.E., 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecological Economics 39, 333-346.

Kumar, P., Esen, S.E., Yashiro, M., 2013. Linking ecosystem services to strategic environmental assessment in development policies. Environmental Impact Assessment Review 40, 75-81.

Lamorgese, L., Geneletti, D., 2013. Sustainability principles in strategic environmental assessment : A framework for analysis and examples from Italian urban planning. Environmental Impact Assessment Review 42, 116-126.

Landsberg, F., Ozment, S., Stickler, M., Henninger, N., Treweek, O., Venn, J., Mock, G., 2011. Ecosystem Services Review for Impact Assessment : Introduction and Guide to Scoping, WRI Working Paper. World Resources Institute, Washington DC.

Laurans, Y., Argaud, F., 1999. Évaluation économique des services rendus par les zones humides à l'homme. Etude de cas : la moyenne vallée de l'Oise. Programme National de Recherche sur les Zones Humides, in : AScA, Paris.

Laurans, Y., Cattan, A., Dubien, I., 1996. Les services rendus par les zones humides à la gestion des eaux : évaluation économique pour le bassin Seine-Normandie, in : Asca. Agence de l'Eau Seine-Normandie, France.

Laurans, Y., Rankovic, A., Billé, R., Pirard, R., Mermet, L., 2013. Use of ecosystem services economic valuation for decision making : Questioning a literature blindspot. Journal of Environmental Management 119, 208-219.
Leonard, J., Le Bissonais, Y., Andrieux, P., Darboux, F., 2009. L'érosion, un acteur majeur de la dégradation des sols et de l'environnement, in : Stengel, P., Bruckler, L., Balesdent, J. (Eds.), Le Sol. Dossier INRA, Edition Quae, Versailles, pp. 128-131.

Lindenmayer, D.B., G., L., 2005. Synthesis : Thresholds in conservation and management. Biological Conservation 124, 351-335.

Lindenmayer, D.B., Fischer, J., Cunningham, R.B., 2005. Native vegetation cover thresholds associated with species responses. Biological Conservation 124, 311-316.

Lonsdorf, E., Kremen, C., Ricketts, T., Winfree, R., Williams, N., Greenleaf, S., 2009. Modelling pollination services across agricultural landscapes. Annals of Botany 103, 1589-1600.

Loomis, J.B., Rosenberger, R.S., 2006. Reducing barriers in future benefit transfers : Needed improvements in primary study design and reporting. Ecological Economics 60, 343-350.

Lovett, G.M., 1994. Atmospheric deposition of nutrients and pollutants in north America : An ecological perspective. Ecological Applications 4, 629-650.

Mancebo Quintana, S., Martín Ramos, B., Casermeiro Martínez, M.A., Otero Pastor, I., 2010. A model for assessing habitat fragmentation caused by new infrastructures in extensive territories - Evaluation of the impact of the Spanish strategic infrastructure and transport plan. Journal of Environmental Management 91, 1087-1096.

MAP/IFN., 2006. Les Indicateurs de Gestion Durable des Forêts Françaises, Paris, p. 148.

McPherson, E.G., Nowak, D., Heisler, G., Grimmpnd, S., Souch, C., Grant, R., Rowntree, R., 1997. Quantifying urban forest structure, function, and value : the Chicago urban forest project. Urban Ecosystems 1, 49-61

Mendoza-Gonzàlez, G., Martínez, M.L., Lithgow, D., Pérez-Maqueo, O., Simonin, P., 2012. Land use change and its effects on the value of ecosystem services along the coast of the Gulf of Mexico. Ecological Economics 82, 23-32.

Meynecke, J.O., Lee, S.Y., N.C., D., 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. Biological Conservation 141, 981-996.

Millenium Ecosystem Assessment, 2005. Ecosystem and well being : A framework for assessment. Island Press Blurb, Washington DC.

Mitchell, M.E.G., Bennett, E.M., Gonzalez, A., 2013. Linking landscape connectivity and ecosystem service provision : current knowledge and research gaps. Ecosystems 16, 894-908.

Montagné, C., Niedzwiedz, A., 2009. Comptes Intégrés économiques et environnementaux de la Forêt en France : Méthodologie et Analyse pour la Période 2000-2006. Rapport final. Juin 2009, Paris.

Muradian, R., 2001. Ecological thresholds : a survey. Ecological Economics 38, 7-24.

Ng, C.N., VXie, Y.J., Yu, X.J., 2013. Integrating landscape connectivity into the evaluation of ecosystem services for biodiversity conservation and its implications for landscape planning. Applied Geography 42, 1-12.

Nicholls, C.I., Parella, M., Altieri, M.A., 2001. The effects of a vegetational corridor on the abundance and dispersal of insect biodiversity within a northern California organic vineyard. Landscape Ecol 16.

Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban Forestry and Urban Greening 4, 115-123.

Opperman, J.J., Luster, R., McKenney, B.A., Roberts, M., Meadows, A.W., 2010. Ecologically functional floodplains : connectivity, flow regime, and scale. Journal of the American Water Resource Association 46, 211-226.

Panagos, P., Meusburger, K., Alewell, C., Montanarella, L., 2012. Soil erodibility estimation using LUCAS point survey data of Europe. Environmental Modelling & Software 30, 143-145.

Partidario, M.R., Gomes, R.C., 2013. Ecosystem services inclusive strategic environmental assessment. Environmental Impact Assessment Review 40, 36-46.

Perrings, C., Pearce, D., 1994. Threshold effects and incentives for the conservation of biodiversity. Environmental & Resource Economics 4, 13-28.

Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E., 2000. Biodiversity conservation at multiple scales : functional sites, landscapes, and networks. BioScience 50, 133-146.

Quinet, A., Baumstark, L., Célestin-Urbain, J., Pouliquen, H., Auverlot, D., Raynard, C., 2009. La valeur tutélaire du carbone. Documentation française, Paris.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, R., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100 Science 287, 1770-1774.

Sanderson, E.W., Jaiteh, M., Levy, M.A., Redford, K.H., Wannebo, A.V., Woolmer, G., 2002 The human footprint and the last of the wild. BioScience 52, 891-904.

Seiler, A., 2002. Effects of Infrastructure on Nature, in : Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J. (Eds.), COST 341 - Habitat Fragmentation due to transportation infrastructure : The European Review. Office for Official Publications of the European Communities, Luxembourg., pp. 31-50.

Suding, K.N., Hobbs, R.J., 2009. Threshold models in restoration and conservation : a developing framework. Trends in Ecology and Evolution 24, 271-279.

Swift, T.L., Hannon, S.J., 2010. Critical thresholds associated with habitat loss : a review of concepts, evidence and applications. Biological Reviews 85, 35-53.

Taylor, P.D., Fahrig, L., Henein, K., Merriam, G., 1993. Connectivity is a vital element of landscape structure. Oikos 68, 571-573.

TEEB, 2010. The Economics of Ecosystems and Biodiversity : Ecological and Economic Foundations. In : Kumar, P. (Ed.), Earthscan, London, Washington.

Thompson, J.D., Mathevet, R., Delanoë, O., Gil-Fourrier, C., Bonnin, M., Cheylan, M., 2011. Ecological solidarity as a conceptual tool for rethinking ecological and social interdependence in conservation policy for protected areas and their surrounding landscape. Comptes Rendus de l'Académie des Sciences, Biologies 334, 412-419.

Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J., 2002. COST 341 -Habitat Fragmentation due to transportation infrastructure : The European Review. Office for Official Publications of the European Communities, Luxembourg.

Turner, R.E., Rabalais, N.N., 2003. Linking landscape and water quality in the Mississippi River basin for 200 years. BioScience 53, 563-572.

van der Zee, D., 1990. The complex relationship between landscape and recreation. Landscape Ecology 4.

Vigiak, O., Sterk, G., Warren, A., Hagen, L.J., 2003. Spatial modeling of wind speed around windbreaks. CATENA 52, 273-288.

Watkiss, P., Holland, M., Hurley, F., Pye, S., 2006. Damage Costs for Air Pollution, in : DEFRA (Ed.). AEA Technology, London, UK.

With, K.A., Gardner, R.H., Turner, M.G., 1997. Landscape connectivity and population distributions in heterogeneous environments. Oikos 78, 151-169.

Xue, D., Tisdell, C., 2001. Valuing ecological functions of biodiversity in Changbaishan Mountain Biosphere Reserve in Northeast China. Biodiversity and Conservation 10, 467-481.

Chapter 4

# Ecosystem services in the evaluation of transport infrastructure projects - an added value?

## Abstract

In this chapter we provide a test of the usefulness that may have ES consideration into the evaluation of transport infrastructure projects. Projects are built and assessed according to two complementary tools, the environmental impact assessment (EIA) and the cost benefit analysis (CBA). We provide an attempt to mainstream ES into EIA and CBA in order to enlarge the scope of spatial planning and to assess the additional information it may bring. Attention is paid to the applicable character of methods to the real legal framework within which EIA and CBA must be conducted. We show that this type of analysis can enlighten and provide guidance at different stages of a transport project: from preliminary studies to the study of the final implementation option. Specifically, this type of analysis can help designing more appropriate environmental measures by expanding the types of impacts assessed, and provide a quantitative assessment of the cost related to the final chosen option. It may help project stakeholders to apprehend the effects on a broader scale instead of staying confined into project boundaries and regulatory checklists.

## Résumé

Dans ce chapitre, nous tentons de montrer l'utilité de la prise en compte des services écosystémiques dans l'évaluation des projets d'infrastructures de transport. Les projets sont élaborés à l'aide de deux outils : l'étude d'impact environnementale et le bilan socio-économique (construit sur les bases d'une analyse coût-avantage). Nous tentons donc d'inclure les services dans ces deux outils de manière à élargir le champ de vision de l'aménagement du territoire, afin de montrer l'utilité d'une telle intégration. Une attention particulière est portée au caractère applicable de l'analyse aux cadres réglementaires actuels entourant l'étude d'impact et le calcul socio-économique. Nous montrons que ce type d'analyse peut éclairer et orienter différentes étapes d'un projet d'infrastructure : des études préliminaires, à l'étude du tracé final. Plus spécifiquement ce type d'étude peut permettre une meilleure identification des mesures d'insertions les plus appropriées, en élargissant le types d'impacts pris en compte et en donnant des indications quantitatives des coûts engendrés par les différentes options d'implémentation. Ceci peut permettre aux parties prenantes du projet de mieux appréhender les différents effets engendrés par le projet, à une plus grande echelle, leur permettant de sortir des strictes frontières du projet et des contrôles réglementaires.

## 4.1 Introduction

The use of Ecosystem Services (ES) concepts to support real-life decision-making processes is still limited (Laurans et al, 2013). Studies assessing ES remain restricted to the illustration of the importance of preserving ecosystems assuming that it may have an indirect influence on decision-making. According to Haines-Young and Potschin (2011), the main reason for why ES are rarely addressed for more practical decision making is that the currently available models (ecological or economic) accounting for ES are becoming increasingly complex but with a little thinking on whether they might be applied to practical cases. The exact opposite reason is also raised to explain this issue: the spatial and temporal dynamics in service provision and its value still have to be better accounted for, to yield more robust results regarding real spatial tradeoffs (Nelson et al, 2013). The last reason put forward is that ES valuation is sometimes perceived as not suited to certain types of ecosystem services, that possess remarkable characteristics and therefore might be considered as invaluable.

Yet, land management decisions still have substantial effects on ecosystems and on the goods and services they provide. By characterizing these services which connect the ecological dimension to the socio-economic world, we begin to understand the whole variety of valuable tradeoffs associated with land use change (Nelson et al, 2013). The inclusion of ES in spatial planning is increasingly viewed as a mean to create a more rational organization of land uses, to balance demands for development with the need to protect the environment (European Commission, 1997). Assessing the impacts of plans, policies or development projects on a wide range of ES can be integrated into more cost-effective policy implementation, where synergies might emerge and yield win-win situations (De Groot et al, 2010; Geneletti, 2013; TEEB, 2010). The ecosystem services based assessment is thus intended to be used to explain tradeoffs that will enable decision-makers to monitor multiple objectives. However, the inclusion of ES might be expected to influence biodiversity and ES level provided that the usefulness of the concept is demonstrated and made operational for practical decision making.

Recent initiatives and projects are developed to counter the lack of practical considerations of ES. In 2012, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was created to provide an interface between the scientific community and policy makers, and build capacity to reinforce the use of ES science and assessments in policy making (Ruckelshaus et al, 2014). Besides, national assessments as the EFESE[1] project in France and other initiatives of large scale ES assessments have been developed increasingly, including the National Ecosystem Assessment in the UK (Bateman et al, 2011), the Natural Capital project (Kareiva et al, 2011), the Valuing the Arc initiative in the Eastern Arc Mountains, Tanzania (Fisher et al, 2011), and ARIES (Bagstad et al, 2013) for mainly regarding USA case studies.

Two very recent papers (Presnall et al, 2014; Mascarenhas et al, 2014) assessed the potential of integrating ES in real-world spatial planning, by interviewing spatial planning stakeholders. Presnall et al (2014), implemented the survey in the case of the U.S. Forest Service National Environmental Policy Act, and surveyed over 500 U.S. Forest Service Professionals. They mainly found that a large part of respondent were unfamiliar with the concept (41%), but that the majority thought that the integration of ecosystem services could be helpful in the process in terms of improved quality of communication and analysis (relevance and efficiency). Negative points underlined the unclear definition of the ES and that practical method and data are lacking, but also that the integration could make already cumbersome documents longer without improving decisions. Finally, a small number of respondents fear that these analysis would make documents more vulnerable to appeal and litigation. Mascarenhas et al, (2014), surveyed stakeholders from the Portuguese regional spatial planning authorities. They found a greater familiarity with the concept and the same feeling of importance of the integration but interestingly that respondent have think also that ES is already rather integrated in existing plans. However they revealed a low knowledge on the main initiatives intended to push ES into the political agenda.

In this chapter we provide a test of the usefulness of the ES inclusion into the evaluation of one of the major drivers of habitat loss and landscape fragmentation that is linear terrestrial transport infrastructure. Projects are build and assessed according to two complementary tools, the environmental impact assessment (EIA) and the cost benefit analysis (CBA). We provide an attempt to mainstream ES into EIA and CBA in order to enlarge the scope of spatial planning and to assess the additional information it may bring. Attention is paid to the application of methods in the real legal framework within which EIA and CBA must be conducted. We show that this type of analysis can be informative and useful at different stages of a transport project: from preliminary studies to the report

[1]EFESE: "Évaluation Française des Écosystèmes et des Services Écosystémiques" is a process led by the Minister of Ecology in France recorded in the national biodiversity strategy.

prepared for the public inquiry. Specifically, this type of analysis can help designing appropriate environmental measures by expanding the types of impacts assessed, and provide a quantitative assessment of the cost related to the final chosen option. It may help project stakeholders to apprehend the effects on a broader scale instead of staying confined into project boundaries and regulatory checklists.

The remainder of the chapter is organized as follows. In Section 4.2, we present the legal framework of transport infrastructure projects in France. This presentation is made in order to illustrate how the legal context influences the most appropriate way of incorporating ES into transport project practices. In Section 4.3, we describe the methodology proposed to include ES concept into this framework. In Section 4.4 results are provided on the case of a real high-speed rail project in France. Section 4.5 discuss the results and the possibility of a systematic integration of ES loss in transport projects evaluation and Section 4.6 conclude.

## 4.2 Legal framework of transport infrastructure projects

This section describes the legal framework within which the ES inclusion might operate. The first subsection describes the legal framework for transport infrastructure Environmental Impact Assessment (EIA) and points out the shaping role it plays on the design of the project implementation options. The following subsection describes the regulatory context in which the CBA must be conducted in the transport sector.

### 4.2.1 Environmental Impact Assessment (EIA) legal framework

The EIA aims at studying the overall consequences of a project in order to (i) identify the least impacting implementation option minimising environmental impacts, (ii) design environmental measures in order to implement to the selected option. The general EIA process is presented in Figure 4.1.

In France, several laws frame the process. Since the 1976 nature protection law (Article 2 of law 76-629 on July 10th, 1976), any infrastructure project has to take into account legal environmental measures articulated in three successive measures: avoidance, mitigation, and compensatory measures. Implementation options have to be designed, first of all, by avoiding, as far as possible, regulated protected areas. Avoidance criteria are diverse, and potentially country-specific, but focus primarily on zones containing endangered

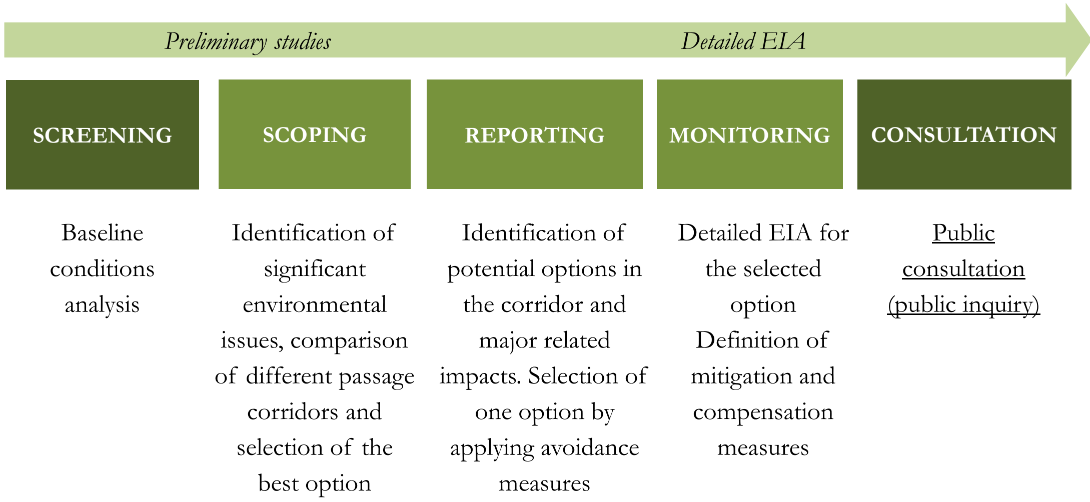



**Figure 4.1:** Environmental Impact Assessment (EIA) process (adapted from Kumar et al, 2013)

species, protected areas, biological reserves, and zones containing key ecological connectivity (identified by regional plans of ecological coherence when the relevant scale is the Region). In France, avoidance measures are based on maps of particular sensitivities of the area in relation to different themes : (i) water resources (flow, streaming due to waterproofing, impact on groundwater resources), (ii) natural environment (tracking sites designated for natural conservation ), (iii) human environment (urban areas, agricultural areas); (iv) landscape and cultural heritage; and finally (v) the impact of noise disturbance and pollution. Sensitivities are represented through qualitative indicators graded from low to strong sensitivities. For instance, in France, sensitivities of natural areas are designated as follows (table 4.1).

Further, project designers have to mitigate residual impacts by adapting the project characteristics in order to correct the identified damages. Specifically mitigating measure are wildlife crossing structures or fauna passages to reduce fragmentation, noise-barriers to reduce noise disturbance, fences (or screens) to reduce fauna mortality. Habitat loss cannot be mitigated, then residual impacts have to be compensated by promoting favourable environmental actions (species habitats creation, ecosystem restoration) which must be (in theory) at least equivalent to the loss incurred.

Avoidance measures are particularly used during the stages of selection of implementation options (scoping and reporting stages). Mitigation measures are designed on the final route option (monitoring and consultation stage, and more precisely specified after the

**Table 4.1:** Example of sensitivities assignment in the EIA

| Very High sensitivity | High sensitivity | Moderated or low sensitivity |
|---|---|---|
| - Natura 2000 (protected areas in the territory of the European Union territory) | - ZNIEFF (Natural Zone of Interest for Ecology, Flora and Fauna sites) Type 1 | - ZNIEFF (Natural Zone of Interest for Ecology, Flora and Fauna sites) Type 2 |
| - Areas protected by order of the prefect | - Areas under the protection of the French "Conservatoire du Littoral" | - Hunting reserves |
| - ZICO (protection of birds species) | - Sensitive Natural Areas protected by departmental policy | |
| - Protected species (species habitats and plants) | - Wetlands and water bodies | |

public inquiry). Both Measures are developed mainly by using geographical information systems and by mapping key components of the landscape. As the project proceeds, the accuracy of potential impact identification increases. The object of the analysis moves from 10-15 km wide passage corridors (during preliminary studies), to 1-5 km spindles, and finally to reference routes throughout the selected spindle of hundred meters wide. Hence, the spatial scale of geographical information systems outputs varies from 1:100 000 (1cm for 1km) or 1:50 000 (depending on the theme) during the first stages of the project process, down to 1:5 000 (1cm for 50m) for the last stages.

Before the consultation stage, an EIA report must be provided, synthesizing all the environmental studies realized from the beginning of the process. It formalizes all the environmental evaluations made at this stage. EIA report must provide justifications for all the choices made (choice of development facilities, selection of the option, and all project related measures). Legally, the content of the EIA report is defined in the article R122-5 of the environmental code. It provides in particular:

- An analysis of the environmental baseline condition, and an identification of the areas likely to be affected by the project.
- A qualitative analysis of all relevant impact on the environment (negative or positive, direct or indirect, temporary or permanent effects) in short, average and long runs.
- A draft of the main implementation options and reasons why (with respect to the effects on the environmental components), the presented project was retained;
- A set of environmental measures designed to tackle the most significant (negative)

impacts on the environment and human health (with a special focus on the expected effectiveness of these measures).

Results are presented through detailed maps, describing the various environmental sensitivities. A general mapping of the sensitivities (in which the accumulation of impacts are based on expert judgment)is then presented and the route option choices are partly justified on its basis. The assessment of potential effects Natura 2000 areas and on protected species has also to appear and be mapped in an individualized way. The environmental study is then submitted to the environmental authority, as codified by article L122-1 of the environmental code. The EIA report and the environmental authority notice are then included in the general document required for public inquiry.

### 4.2.2 Cost-Benefit Analysis (CBA) legal framework

***General introduction to the CBA***

The Cost-Benefit Analysis (CBA) compares the monetary values associated to positive and negative effects of the project. The CBA enable decision-makers to determine if a project is acceptable from an economic and social point of view (opportunity of the project), and to what extent (profitability of the project). In order for different effects to be co-measurable, they must be valued in common units. The common units in CBA is money, and as argued by Hanley and Spash (1993), this is merely a device of convenience rather than an implicit statement that only money is important.

Markets generate the relative value of all traded goods and services as relative prices. Prices are therefore very useful in comparing different effects, but they also give some indications of goods and services relative scarcity. The CBA should be carried in real terms (correcting for inflation related distortions), and then the analyst should convert nominal values into real (relative) prices by using prices indexes. However, in some cases market prices are a bad indicator of the marginal social cost and of the marginal social benefit as it is obviously the case in the absence of market. In this case, unpriced impacts (externalities) are assessed by using non-market valuation techniques. The CBA process can be decomposed in different stages as described in Figure 4.2.

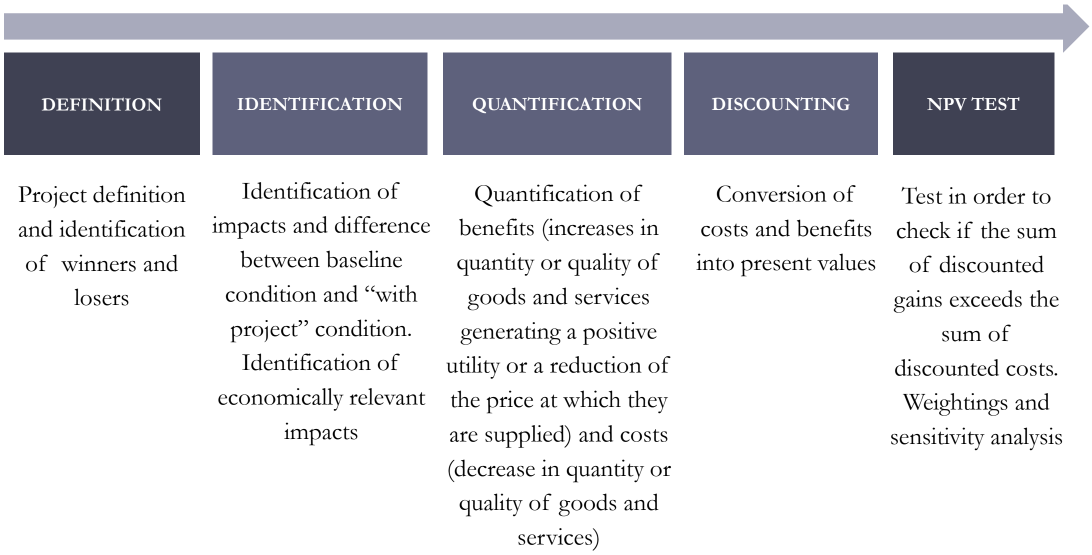



**Figure 4.2:** Cost-Benefit Analysis (CBA) process

***The CBA in the transportation sector***

In the French transportation sector, CBA can be used at two key stages of the project: during the public debate in order to discuss the general opportunity and features of the project (at this stage, the evaluation is only indicative), and, later in order to prepare the public inquiry. The legislation does not impose constraints relatively to the socioeconomic evaluation during the public debate, but the 2005 Framework Instruction (called the Robien Instruction, 2005), stresses the need for a first socioeconomic evaluation at this stage, which includes at least a cost estimate (regarding construction and exploitation), the main advantages of the project, expected negatives effects, the identification of the various groups of beneficiaries, and some information on project funding.

The economic calculation is particularly useful in the current context of scarce public fund. Under a stricter budgetary constraint, the justification of the projects is of growing importance. However the CBA exercises remains widely criticized partly because the assessment is carried out in a partial equilibrium framework. Thus, since the hypothesis of pure and perfect competition is not realistic, it would be necessary to amend the current socioeconomic calculations by accounting for market imperfections (Quinet, 2013). A mean to improve computations is to widen the consideration of the effects (and more particularly external effects) of transport projects.

In most European countries, the socioeconomic assessments tends to widen considerations towards externalities. To quote some recent improvements in the evaluation process have considered agglomeration externalities in the case of United Kingdom, market power in the United Kingdom and Netherlands, or employments created or maintained in Germany (Quinet, 2013). Environmental externalities are receiving increasing attention. In France, the inclusion of externalities as described in Boiteux I and II reports (noise pollution, atmospheric pollution due to the functioning infrastructure, greenhouse gas emissions) was the first formal integration of environmental effects in socioeconomic assessments.

***Scenario and baseline condition for the project evaluation***

To monetize the impact of the infrastructure project, a scenario must be defined regarding the macro-economic framework of the project. Specifically it must provide the evolution trajectories of the national parameters common to all sectors over the project life span period: demographic and macroeconomic indicators (GDP per capita, final household's consumption, population demographic evolution), prices of goods and services most directly concerned (imported energies, wages, interest rate, and others). This implies to take a stand on the general trends of macroeconomic , environmental and energy policies. The scenario must also specify the data related to the concerned sector and the evolution of several supply and demand. By accounting for the medium and long-term uncertainties in several domains, multiple alternative scenarios can be considered.

Finally, transport projects are always compared to a baseline scenario where the project is not being implemented (business as usual scenario), still accounting for other existing or planned transport infrastructure. The effect of the project is assessed as the difference between both scenarios.

***Impacts taken into account for in the current CBA***

The infrastructure has different impacts on actors. For a high speed rail project, the actors are related to (i) the rail transportation sector (managers, organizing authorities and carriers), (ii) other modes of transport such as road administrators (states, departments, municipalities and highway companies) or airline companies, (iii) the users of the mode, (iv) the Government and finally (v) the other parties subject to external effects (third parties). Different regulatory costs and benefits have to be accounted for with respect to these categories of stakeholders: they are summarized in Table 4.2.

**Table 4.2:** Regulatory costs and benefits of a high-speed rail infrastructure project

| **Benefits** | **Costs** |
|---|---|
| *Rail transportation actors (infrastructure administrators and carriers)*<br>- Tickets sales (carriers)<br>- Fees (infrastructure administrators) | *Rail transportation actors (infrastructure administrators and carriers)*<br>- Fees (carriers)<br>- Part of the investment costs (administrators)<br>- Functioning costs, marketing costs renewal costs and rolling stock investment (carriers)<br>- Maintenance costs (infrastructure administrators)<br>- renewal costs (infrastructure administrators) |
| *Other modes actors (airline companies, road administrators)*<br>- Functioning costs (both)<br>- Maintenance and operating costs (both)<br>- Avoided taxes and fees (both) | *Other transport modes actors (airline companies, road administrators)*<br>- Tickets sales (airline companies)<br>- Tolls receipts (highways) |
| *Users*<br>- Savings on plane tickets and tolls<br>- Differential cost of use of car passengers<br>- Time gains * | *Users*<br>Train tickets expenses |
| *Government (public sector)*<br>- Value added tax on train tickets<br>- Others taxes | *Government (public sector)*<br>- Part of the investment costs<br>- Value added tax on airline tickets and tolls<br>- Tax on use of passengers cars (value added tax and tax on domestic tax on petroleum products) |
| *Third parties*<br>- Atmospheric pollution reduction *<br>- Traffic congestion reduction *<br>- Road insecurity reduction *<br>- Noise effects reduction *<br>- Greenhouse gases emission reduction * | *Third parties* |
| ** Monetized with non-market valuation methods* | |

The negative impacts (costs) are mainly investment costs, investments in rolling stock, and operating expenses (cost of renewal, differential maintenance costs, differential functioning and marketing costs). Costs can be converted into benefits, and conversely, depending on the type of infrastructure. For instance, regarding road infrastructure projects, external effects on third parties such as atmospheric pollution of the functioning infrastructure, noise pollution, congestion or road insecurity are accounted as costs. For railroad projects, these external effects are considered as benefits because of the modal transfer of the road users in railroad mode users, the rail mode being less impactful for all negative environmental external effects compared to other modes.

The benefits resulting from the infrastructure are mainly the revenues generated by the use of the infrastructure (revenues from train tickets) and the time saved by travelers. As explained above, in a railroad project the change in external effects that may be considered as benefits are the reduction in the traffic congestion, a safer road access for the remaining users, the reduction in local pollution, in noise and greenhouse gas effects (global pollution) due to lower road and air traffics. The non-market methodologies used to assess the value of environmental external effects that must be accounted for in transport projects are defined by the Boiteux report (2001) (see Appendix C for a brief description), and became mandatory since the Framework Instruction (Robien, 2005).

***Infrastructure life span and public time discount rate***

Infrastructure project assessment is usually made over a 50-years period for railroad projects and over a 100-years period for road projects. Over the project life span, flows of costs and benefits are discounted. Public sector projects in France use regulatory public time discount rate as defined in the Lebègue et al (2005) report (transcribed onto the Robien Instruction, 2005). The public discount rate is unique and applies in a uniform way to all public projects considered and to all business sectors (as far as policy choices are concerned). Moving away from this principle would lead to accept systematic inconsistency in the allocation of the public resources.

The expression of the public discount rate ($r$), which is a risk free discount rate, has two terms:

$$r = \delta + \gamma\mu,$$

where the first term ($\delta$) is the rate of pure time preference, denoting the price that agents (producer or consumer) grant to time and modeling their impatience (or it may be interpreted as the "probability to die" at time each period). Indeed, agents manifest in their consumption or savings behaviour preference for the present: an "immediate pleasure" is generally preferred to the same "pleasure" in the future. This impatience effect is generally estimated on the basis of the interest rate which would be required by the households to postpone their consumption.

The second term ($\gamma\mu$) is a term reflecting a wealth effect, corresponding to the product of the households' anticipations regarding the growth of their consumption, and the absolute value of the elasticity of marginal utility (the decrease of the marginal satisfaction derived from consumption).

If the income growth is certain, it is preferable for the consumer to consume more today, anticipating the growth: then the consumer will not make efforts to improve a future that he anticipate to be better than present. If we consider now that the consumption marginal utility is decreasing with the level of consumption, the fact that this level is increasing with time implies that agents will prefer to consume when the level of utility is higher, that is when they are less wealthy.

The preference intensity is modeled using ($\gamma$) the elasticity of marginal utility of consumption. The higher $\gamma$ is, the lower agents will postpone their consumption. The anticipations on growth result in the same effect: the higher the anticipations are, the more households are prone to consume today. The same effect applies for successive generations. In a growing economy, wealth effect drives the current generations to make fewer efforts when future generations are distant in time, because investing for the future in a growing economy impoverish the present generations for the benefit of the future generations relatively richer because of the growth. Finally, $\mu$ represent the anticipated income growth per capita.

The Lebègue et al (2005) report set the different regulatory values composing the discount rate[2], and its evolution in the long run. The value of the rate of pure time preference $\delta$ is fixed at 1%, and $\gamma$, the elasticity of consumption marginal utility, is fixed at 2 (referring to an INSEE study conducted for the report, which estimates the value of $\gamma$ between 1.8 and 2.35). The growth rate per capita $\mu$ is set at 1.5 %.

In practice, the Lebègue et al (2005) report accounts for an uncertainty on the trend of the consumption growth, by introducing a precaution effect. It was thus decided to assume that the economic growth per capita was equal to 2 % with a probability 2/3, or 0.5% with probability 1/3. This has no impact on the short term discount rate staying at 4%, however this uncertainty impact the long terms horizons and the discount rate is then

[2]However they suggests that periodical revisions must be conducted to stay in accordance with the major macroeconomic indicators: economic growth, interest rate evolution, demographic variables, workforce profile, etc.

decreasing in time, the structure per terms being decreasing. For very long time horizons, the rate is assessed to converge to 3% from 100 years and around 2 % to preserves some value for the very distant future (500 years). For simplification, the Framework instruction (Robien, 2005) prescribes to use a 4% discount rate regarding development projects in the first 30-years, then a 3.5% rate between 30 and 50 years, and finally a 3% rate above 50 years.

Although the public discount rate for infrastructure projects is set and fixed by regulation, we can highlight (in a nutshell, because it is not the purpose of this analysis) some issues debated in the literature. The first issue concerns ethical responsibility to future generations. The responsibility may be captured, at least partially, by the term $\delta$, the rate of pure time preference. Although there is still considerable disagreement among economists, a strong case may be made that $\delta$ should be close to zero, reflecting the idea that there is no reason to put a lower value on the well-being of future generations (Gowdy et al, 2010). The value of the income growth per capita $\mu$, because it is concerned with our expectations about how well future generations will be able to deal with the problems created by the current one. Indeed, the term $\mu$ represents all the productivity factors: when natural capital is included, several studies highlight that $\mu$ can become negative, that is, current economic growth is maintained by drawing down the natural capital common to the various generations (Gowdy et al, 2010).

The second issue is related to the long run evolution of the discount rate. Several theoretical and empirical arguments justifying a time-declining approach to discounting over a long time horizon have been provided in the last years, and several contributions have critically reviewed this issue (Oxera, 2002; Pearce et al, 2003; Hepburn, 2007, among others). The theoretical and empirical arguments regarding a time-declining discount rate are mainly related to: (i) the uncertainty about the future state of the world, about the future path of the discount rate (Weitzman, 1998, 2001), the future growth rate of consumption (Gollier, 2002, 2010) ; (ii) the sustainability and intergenerational equity issues that are explicitly taken into account in order to avoid the dictatorship of one generation (present or future) over the others (Chichilnisky, 1997); and (iii) the increasing experimental evidence that individuals discount hyperbolically when making intertemporal choices (Weitzman, 2001; Gowdy, 2010). Specifically, people discount the value of delayed consumption more in the immediate future as opposed to the distant future (Weitzman, 2001; Settle and Shögren, 2004).

#### ***Non-marketed effects evolution***

Although the different project effects are discounted over time, the legal framework planned the evolution of relative prices of non-marketed effects to be used in the development projects assessments. The evolutions of relative prices are given in the Robien Instruction (2005) and based on the Boiteux II (2001) and Quinet (2009) reports. According to these reports, the value of time savings should evolve yearly for urban and intercity users as a function of the growth of the GDP per capita (with an elasticity of 0.7, this elasticity is described on the literature as varying from 0.5 to 1). The value of time saving in the case of the freight transport should evolve at 2/3 of the growth of real GDP per capita. Noise and "life" values should evolve at the same rate as the GDP per capita. The carbon value progresses as described in the Quinet (2009) report (at 5.8% before year 2030 to reach the value of the time discount rate after year 2030). For local pollutions, the evolution depends on (1) pollutant emissions (assumed to decrease due to the anticipated technical progress: -5.5% for light-weight vehicles; and -6.5% for heavier vehicles such as trucks and buses), (2) the life value (increasing with GDP per capita).

#### ***Acceptance Criteria***

For a given project, various criteria are generally used. The first one is the Net Present Value (NPV) of the project choice decision, defined as the difference between benefits and costs aggregated (for all actors) over the entire project life span. The aggregation of flows generated in different dates is captured using the discount rate ($r$) defined above. The expression of the NPV of a given infrastructure (denoted $i$) expressed in real terms at year t = 0 is thus:

$$NPV_i = \sum_{t=0}^{t} \frac{B_t - C_t}{(1+r)^t} + \frac{RV}{(1+r)^{T+1}}$$

In this expression $B_t$ denotes the annual benefit flow (expected to beging for the functioning infrastructure) and $C_t$ the annual cost flow (where the budgetary funds are multiplied by the opportunity costs of public funds), while t = 0 denotes the year zero of the project, $C_0$ the year zero construction cost, RV the residual value of the project and T the project life span. RV reflects the fact that some investments have not reached their life expectancy at the end of the project life span, either because they were renewed during the period or because they have a longer life expectancy. Hence, these investments are not obsolete at the end of the period and are taken into account through a residual value introduced at the end of the assessment period, which is captured on the basis of a notional resale price.

The socioeconomic NPV is defined for the collectivity and takes into account, besides the financial flows, the non-market flows of costs and benefits which have been monetized. The higher the NPV is, the more the situation where the project is implemented can be evaluated as preferable to the baseline condition (criterion based on the Kaldor-Hicks principle in neoclassical welfare economics). The criterion for project acceptance is a non-negative NPV: in that case the project is deemed to result in an improvement in social welfare. But it does not ensure at all that there is no other more profitable project. In particular, this statement applies to the implementation date, because the same project implemented in two different dates will constitutes two alternative variants.

Two supplementary indicators can be analyzed: the internal rate of return, and the benefit-cost ratio. - The internal rate of return is the interest rate which, if used as discount rate for the project, would yield a NPV of zero. It allows for an estimate of the project profitability without referring to a particular discount rate. The NPV being generally a decreasing function of the discount rate (Hanley and Spash, 1993), since most of the costs are borne during the first years and the benefits are generally larger in the following periods, the project should be realized only when the internal rate of return is larger than the public discount rate. - The benefit-cost ratio is measure of the NPV per euro invested, the decision rule being to implement the project if the ratio is larger than one. The same calculation is applied for public euro invested, the decision rule being that every public euro invested in the project has to generate a benefit of at least 30% of the public expense (Robien Instruction, 2005).

Different weights can be applied within the NPV function framework to correct inequities induced by conventional CBA (which is consistent with individual preferences but give priority to richer as they are likely to have higher willingness to pay). Society then may put a higher weight more importance on losses incurred by poor groups than on those incurred by rich groups. However issues such as the definition of weights or of groups constitutes the mains reason why this weighting procedure is rarely used for public decision making.

Sensitivity analysis is finally required to check the relative sensitivity of computations with respect to data input. Such an analysis reveals the degree of robustness of the results with respect to uncertainty about future physical flows, future relative values, discount rate and project life span.

### 4.2.3 Decision on project implementation

At the end of the process, the document prepared for public inquiry is subject to public consultation. It includes the EIA study, the environmental authority notice and the cost-benefit analysis. The public inquiry takes place under the supervision of a superintendent asked to collect public opinion and to provide a notice regarding the pursuit of the project (favorable, favorable with condition or unfavorable). If the project is considered as socially favorable it is designated as "public utility project", and the final implementation option is validated.

## 4.3 Toward a consideration of ecosystem services in the evaluation of transport infrastructure projects

This section begins by outlining the main reasons to include ecosystem services (ES) considerations in the evaluation of transport infrastructure projects. Further, the section describes a methodology proposal for the ES loss inclusion, build upon the results of the preceding chapters, that is adapted to current legal framework surrounding transport infrastructure projects.

### 4.3.1 Main reasons for integrating ecosystem services in EIA and CBA

The inclusion of considerations to ES in transport infrastructure project assessment would be justified on several grounds. The first general reason is statutory put forward in the Quinet (2013) report. The Convention on Biological Diversity (CBD) adopted, in 2010, in Nagoya, a strategic plan to limit the loss of biodiversity and of ES by 2020. One of the main objectives is to reduce the conversion of natural and semi-natural ecosystems (Target 5 of the Aïchi biodiversity targets):

> "By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced."

France is a signatory of this convention, and one of its priorities consists in the avoidance of biodiversity and of ES net loss. Infrastructure constitutes an important driver of habitat loss and fragmentation, which in turn result in the loss of ES. The EIA and the CBA processes that aims at evaluate transport infrastructures should hence integrate avoidance of key ES loss as a new objective.

The second general reason is that the ES approach does enable to identify impacts that had never been characterized before (nor even mentioned). Table 4.3 lists the ES that may be, at least partially, accounted for in current transport projects EIAs, those that are not considered even indirectly, and the ES we will account for in order to provide additional information on impacts on ecological functions and human benefits. We can see that some ES can be potentially partially accounted for through existing environmental measures (passageway construction, landscaping and acoustic insertion, afforestation). However, given that measures are not assessed through an ecosystem-service based approach the restoration of the service is not an objective, thus impacts on ES provision are likely to be maintained. More precisely, consider a peri-urban forest that previously regulated the air quality of a city, that is destroyed and compensated by afforestation in another location far from the city, then the service can be considered as lost. Indeed, in this case the "peri-urban" variable is not automatically taken into account. In the case of the CBA no impact of the infrastructure construction on ES is taken into account (nor approached) in the process.

Yet, ES approach provide a way to understand the most important interactions between society and the environment, identifying issues that matter for the decision-making context. It is a way to identify the significant socioeconomic losses that are due to environmental impacts (Landsberg et al, 2011). As Ruckelshaus et al (2014, p. 8) argue:

> "Having the ability to follow biophysical ecosystem service estimates through to economic values has proven to be an important conceptual advance that has opened many decision makers to discussions they previously did not consider".

Further, this approach account for multi-scaled impacts because ES are supplied and demanded at local, regional, national and global scales (see Chapter 1). Finally, it allows for a better identification and improves the specification of direct and indirect impacts because it offers a more functional approach (see Chapter 3). This could contribute to widen the scopes of both EIA and CBA processes.

Increasing attention is paid to the ecosystem services-inclusive EIA, with particular emphasis on spatial planning (Baker et al, 2013; Geneletti, 2011; Geneletti, 2013; Kumar et

**Table 4.3:** Impacted ES that can be at least partially accounted for in the French EIA framework (A: avoidance measures; M: mitigation measures; C: compensation measures) and ES accounted for in this analysis

| Service | Function | Benefit | EIA | This study |
|---|---|---|---|---|
| Picking products | Provision of picking food (berries, mushrooms), or ornamental products | Enjoyment of free natural picking products | C | x |
| Raw materials | Provision of natural raw materials | Free natural raw materials (firewood, fodder) | C | x |
| Freshwater | Filtering, retention and storage of fresh water | Provision of water for consumptive use: drinking, irrigation | A | |
| Air quality | Pollutant and particles partial removal by vegetation | Maintenance of good air quality.(health) | C | x |
| Local climate | windbreak effect performed by hedges, cooling effect for urban areas | Crops yields protection | C | x |
| Global climate | Carbon storage performed by soil and vegetation | Maintenance of a favorable climate for human habitation, health, cultivation | | x |
| Flood | Rainfall interception by vegetation, or wetlands buffering against floods | Protection of housings, materials, crops (avoidance of damages) | C | x |
| Water flows | River flows support during the low water period | Water availability for crops or other usage in low water period | C | x |
| Waste treatment | Pollution control, detoxification, filtering of dust particles | Maintenance of a good water quality (abatement of water pollution costs) | | |
| Erosion | Sediment retention by vegetation | Maintenance of arable land | | x |
| Pollination | Suitable living space to allow the abundance of wild pollinators (nesting and breeding) | Crops pollination, production | | x |
| Biological control | Control of pests and diseases, Reduction of herbivory (crop damage) | Crops protection | | x |
| Aesthetic | Attractive landscape features | Enjoyment of scenery | A | |
| Recreation | Variety in landscapes with (potential) recreational uses | Travel to natural ecosystems for eco-tourism,outdoor sports, etc. | M | x |
| Hunting | Suitable living space (habitats and migration) for game animals | Outdoor recreation by hunting wild game fauna | M | x |
| Fishing | Suitable living space for habitats for fishs | Outdoor recreation by fishing | M | x |

al, 2013). Several organizations now promote ES inclusion in general environmental assessment by producing briefings, guidance or other forms of support in the area of ecosystem services (IEMA, 2012; OECD, 2010; WRI, 2011). However the specific case of transport infrastructure is not described. The ES inclusion we propose can guide at all the stages of the EIA process of transport infrastructure projects. First, a quick-scan of key ES can be provided for preliminary environmental studies (*i.e.* screening and scoping stage) to map baseline conditions of the area considered. This can be made through indicators representing key ES in the area. This representation may give information for preliminary avoidance measures by identifying relevant ES on which the targeted project and relevant stakeholders depend (Kumar et al, 2013). Further, it may constitute a new discrimination criterion and a new environmentally-based rationale during the comparison of implementation options (reporting stage). This may result in a more holistic mitigation strategy focusing on ecosystem functions and limits, helping ensure the sustainability of project-related outcomes. This can result in a stronger framework for avoidance and mitigation strategy and in a better understanding in management impacts. For the monitoring stage, the ES valuation would allow for a more accurate comparison of environmental measures to be implemented according to their relative costs and benefits, which in turn would better inform the choice of the appropriate measures. Then, the environmental studies and the economic valuation eventually become complementary: the environmental study suggest effective environmental measure; and the economic valuation identifies the most cost-effective one (Quinet, 2013). Finally it can answer to a public concern related to transport projects (EIA consultation stage). Indeed, a majority of the reports on public debate highlight that the chief public concerns relate primarily to issues of landscape planning, environmental management, public health and project cost, in particular for the case of large-scale projects. The analysis bring forward additional information enlightening not only decisions makers but also the general public at every stage of the process.

For the contribution to the CBA process, target 2 of the Aïchi biodiversity targets prescribes:

> "By 2020, at the latest, biodiversity values have been integrated into national and local development strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems".

Expressing the ES loss in monetary terms make possible to account for economic losses induced by the final option of the project, and contrast it (in the socioeconomic calculation) to other external effects of the project. Even though valuation techniques face various challenges to deal especially with uncertainty, irreversibility and resilience, the estimates derived can thus be used to provide policy makers with an order of magnitude of the loss, and thus of information on the importance of ES on a regular basis (Kumar et al, 2013).

### 4.3.2 Methodological options for integration

***Ecosystem service indicator for preliminary environmental studies***

Decision-makers usually ask for easy-to-use decision support tools that can be quickly incorporated into science-policy processes (Ruckelshaus et al, 2014). The process of analyzing multiple ecosystem functions in different scenario conditions involves the use of very high amounts of information and data. It is unclear to what extent these methods may be used in actual spatial planning settings. The open challenge consists in developing ways to provide decision makers with information in a manageable way, possibly by creating new indicators that consist in combinations of ecosystem services indicators and of indicators traditionally used for land-use planning (Geneletti, 2011; Burkhard et al, 2012). Such indicators would constitute practical instrument that would provide a quick-scan of the initial features of an area via a characterization of related ES supply and demand. One simple way to design this first tool is to map the ordinary ES "hotspots" in term of presence and importance of their supply and demand in the area considered. An advantage of this approach is that it can be automatically incorporated into the existing environmental vulnerability maps to illustrate baseline conditions for the screening and scoping stage of the EIA process. Furthermore, it provides information without being as costly and time consuming than a more detailed spatial assessment of ES loss.

In order to build this indicator we firstly map ES presence derived from land use-land cover data, according to the analysis provided in Chapter 3 (in particular Figure 3.2 in the chapter 3), to illustrate areas that are likely to exhibit a particular loss if the project is implemented. ES presence is represented trough a hierarchical typology of terrestrial ecosystems according to the number of services they potentially supply, taking into account the various spatial conditions for ES presence. We apply a score equal to 1 when the service is supplied unconditionally. Where the ES is provided only under spatial conditions by an ecosystem type, we apply the score 1 if the spatial condition is fulfilled (*e.g.* water flows regulation are only performed by wetlands related to a water system; or recreation services are demanded only if the recreational site is situated at proximity to a city), and 0 otherwise. In order to take into account hunting recreation, erosion prevention and pollination (not associated to a particular ecosystem type), we apply a score of 1 when the layers calculated by using Optiflux and InVEST models identify the service presence, and 0 otherwise.

Besides, the indicator provides information on the different ecosystems abilities to supply ES, considering supply and demand levels at the beginning of the project. The different land cover types' abilities to provide particular ES in the area is assessed (on a scale from 0 to 1) on the basis of biophysical data for provisioning and regulating services and economic values for recreation services assessed in the Chapter 3. A zero score means that the ability considered of a particular land cover type is not relevant to supply the selected ecosystem service. A score of 1 represent the maximum ability of ecosystems to supply the service in the area. Intermediate levels are estimated as a percentage of the maximum level of ES provision (by different ecosystems). This estimates requires a preliminary assessment of ES levels in terms of biophysical quantities.

The combined indicator is the multiplication between the first and the second indicator. A score of 1 indicates a service (potential) presence, and a relative high ability of the area to provide ES. The aggregated scores (for different services) serve as indicators of the ability of different area to provide multiple ES.

***Temporal trends of ES values***

The integration of ES in other stages of the EIA and on the CBA is made by using monetary values of ES. When accounting for ES loss in monetary terms we need to specify assumptions made on the temporal trends of ES values. The choice of the discount factor is generally critical and even more when it comes to ecosystems services and biodiversity because of ethical concerns that are brought into play.

To tackle the issue of loss irreversibility, environmental economics has a long tradition in using the Krutilla and Fisher (1975) model, which explicitly deal with the irreversible nature of development projects. They suggest that development benefits will decrease annually due to technical progress while preservation benefits will increase annually. Specifically, Krutilla and Fisher (1975) argue that development benefits will fall over time especially regarding projects for which benefits are measured as cost savings with respect to alternative technologies (as it is the case here). Indeed, in transport infrastructure projects the technology is "frozen in" while other technologies (those not involved in the project) continue to improve; as such cost savings related to other technologies are likely to decrease. Environmental benefits are also likely to grow over time. Hanley and Spash (1993) state at least three reasons explaining this growth of environmental relative prices. First, because of increasing relative scarcity, as natural (and semi-natural) areas declines (MEA, 2005) and have a limited substitutability, remaining areas becomes more valuable and the willingness to pay (WTP) for their preservation increases due to the law of diminishing marginal utility. Second, because information on the role of ecosystems for human well-being is constantly increasing, people are getting better informed and more likely to have higher WTP for natural resources. The final reason is that an increased material prosperity may augment WTP for scarce non-market goods. A dual discount rate is proposed, one for the general development costs and benefits, and one for the environmental ones.

Since the Krutilla and Fisher (1975) contribution, many authors have suggested to act on the evolution of the relative prices for environmental goods rather than on the discount rate itself (Pearce et al, 2006;. Hoel and Sterner, 2007; Sterner and Perrson, 2008; Lebègue et al, 2005; Chevassus-au-Louis et al, 2009). Thus to assess environmental benefits, two effects influence the values: the social discount rate and the increase of environmental goods values compared to the general price level (related to manufactured goods values). The rationale of the approach is that the projected income growth is expected to increase WTP for preservation over time, this related to growing scarcity and non-substitutability. This is accounted for by assuming a positive income elasticity of WTP for environmental quality improvement that is a measure of how willingness to pay is affected by changes in income (Pearce et al, 2006).

At that point, the main difficulty is related to the modeling of the future for non-market environmental goods and to the way to take into account (in a relevant way) irreversible situations. To do so, it would be necessary to determine the share of utility derived from non-market goods, the elasticity of substitution, as well as the level of non-market impacts (Sterner and Persson, 2008). Some recommendations on the specification of income elasticity of WTP have been made in the literature. For instance, Pearce (2006) provides empirical estimates and suggests using a value of elasticity for environmental goods that is lower than one, and being in the interval [0.3-0.7]. This is supported by Kriström and Riera (1996), Hökby and Söderqvist (2003), or Jacobsen and Hanley (2009), which all support, that the value income elasticity of WTP lies within interval [0-1]. WTP for environmental goods then grows less than proportionally with respect to income (which are consequently necessary goods). However, there is still an active debate about whether services can be classified as necessary goods or luxury goods (cf. Hökby and Söderqvist 2003; Kriström and Riera 1996; Pearce, 1980).

We use specific values for income elasticity of WTP parameters. This assumptions have to be tested by sensitivity analysis in order to access the sensitivity of the results to these assumptions. By making these assumptions of evolution of relative prices (expressed through income elasticity of WTP) we want to illustrate the fact that some ecosystem services are inherently very hard to replace. It can also be justified by the fact that the time-span of environmental damage is likely to be longer than that of other costs and benefits. To specify the evolution of ES values on time, we retain the principle to characterize this evolution according to 4 hypothetical cases, already advanced in the Chevassus-au-Louis et al (2009) report. The evolution is defined as a function of long term irreversibility and substitutability of the services supplied. This definition is as follows:

- Case 0: no evolution over time;
- Case 1: irreversible losses of components of technically substitutable ecosystem services;
- Case 2: losses of irreplaceable elements of biodiversity but for which the conceivable consequences do not threaten the survival of human society;
- Case 3: losses of essential components of biodiversity, for which the consequences are unpredictable and may threaten the survival of our societies.

The main difficulty is then related to the practical characterization of these cases. With these cases, the evolution of relative prices depends on: the importance of the services in the future (because of their scarcity), the substitutability of the environmental goods, the irreversibility of their loss, and the potential impact of policy option on ES relative importance. We first assume that the political objectives, such as the stabilization of the biodiversity level by 2020, constitute a reference of importance. Then every type of ES has a relative price evolving over time, and subsequently Case 0 "no evolution" does not apply to ES loss, even for ecosystems with no conservation status. In case 1, services are assumed to be technically substitutable, the evolution of the relative prices is interpreted as the loss of an option value for which an income elasticity of 0.3 is applied. In case 2, we assume that the income elasticity of WTP is higher than that in case 1, and we use an elasticity of 0.5. This applies to the freshwater supply service, local climate, waste treatment, pollination and recreation services. A low elasticity of substitution is then assumed in this case which implies a larger increase in the relative price of environmental quality because the well-being provided is not easily compensated for by increases in consumption levels of other goods. The case 3 rely to the disqualification of economic valuation in a CBA perspective, the costs being infinite, the loss should be strictly avoided. Assumptions on elasticities of WTP related to income are resumed in the Table 4.4.

**Table 4.4:** Values of income elasticities of WTP retained per ecosystem services

| Ecosystem service | Situation considered | Income elasticity considered |
|---|---|---|
| Picking products (food and raw) | Situation 1 | 0.3 |
| Freshwater | Situation 2 | 0.5 |
| Air quality regulation | Situation 1 | 0.3 |
| Local climate regulation | Situation 2 | 0.5 |
| Global climate regulation | - | - |
| Flood prevention | Situation 1 | 0.3 |
| Regulation of water flows | Situation 1 | 0.3 |
| Waste treatments | Situation 2 | 0.5 |
| Erosion prevention | Situation 1 | 0.3 |
| Wild pollination | Situation 2 | 0.5 |
| Biological control | Situation 1 | 0.3 |
| Recreation | Situation 2 | 0.5 |
| Hunting recreation | Situation 2 | 0.5 |
| Fishing recreation | Situation 1 | 0.5 |

Values then evolve annually according to the factor $(1 + e_k.\mu)^t$ , where $e_k$ denotes the income elasticity of WTP for the ecosystem service $k$ considered , $\mu$ denotes the real income rate per capita. $\mu$ is assumed to increase at 1.5% per year: this assumption was retained by the Lebègue report (2006), and is consistent with real-world estimates: the OECD published a recent report (November 2012) about the perspectives of the world economy and the evolution of the GDP per capita has been assessed to be in average 1.6% over the period 2011-2030 and 1.3% over the period 2011-2060. The evolution of the relative price of environmental goods and services is quite close from the prescription of the Chevassus-au-Louis et al (2009) report, suggesting an evolution of 1% per year. The factor of evolution is however slightly lower in our case, the factor suggested by Chevassus-au-Louis et al (2009) being equal to $(1,01)^t$, and our factor being equal to $(1,0045)^t$ (for an elasticity value of 0.3) and $(1,0075)^t$ (for an elasticity value of 0.5) per year.

## 4.4 Case study

We apply the methodology to a case of a high speed rail project in western France[3]. The analysis of ES integration is made at different key moments of the project given the data available at these different moments. Specifically, the analysis is made first for the stage of preliminary environmental studies within different passage corridors. Then the test is made for the detailed EIA, when the different implementation options of the project are compared (reporting stage). The analysis is performed on two different zones of the project, the first zone having three optional routes and the second zone two optional routes. We also test the usefulness of ES considerations to monitor a mitigation measure (applied at the monitoring stage). Finally, the methodology is applied during the CBA process assessing the effects of the final option chosen for the project.

### 4.4.1 ES indicators for the preliminary environmental study

Indicators are calculated for all ecosystems in the area, aggregated over all services assessed, and mapped to illustrate baseline conditions in the selected corridor (see Figure 4.3). The first indicator map represent the presence of ES according to the ecosystem types and the spatial conditions (Figure 4.3a). The map highlights that at a maximum of ten services are potentially supplied in the area. The combination with the abundance of services will help to characterize the importance of ES in the area. The map representing the combination of services is represented in the Figure 4.3b. These maps thus provide a quick-scan of the main "hotspots" in terms of ES presence and abundance that may help designing first avoidance measures. Both maps already reveals that the south option (at the beginning of the corridor) deliver a lower amount of ES and in a lower abundance than the other ones, and that it may be the best option regarding ES.

Preliminary environmental studies are of crucial importance for avoidance measures. Avoidance criteria in France seeks at preserving (wherever possible) natural areas with a high functional properties and regional ecological corridors. Therefore at this stage, it can be helpful to locate principal areas delivering ES, giving additional information on ecological functions and on landscape connectivity underlying the ES supply. This type of analysis can widen the scope of the assessment of alternative routes, and may constitute a supplementary way to promote the conservation of areas of environmental interest.

[3]It can be noted that the analysis developed here will not have an impact on the project, which already passed the public inquiry and receive a favourable notice for the pursuit of the project.

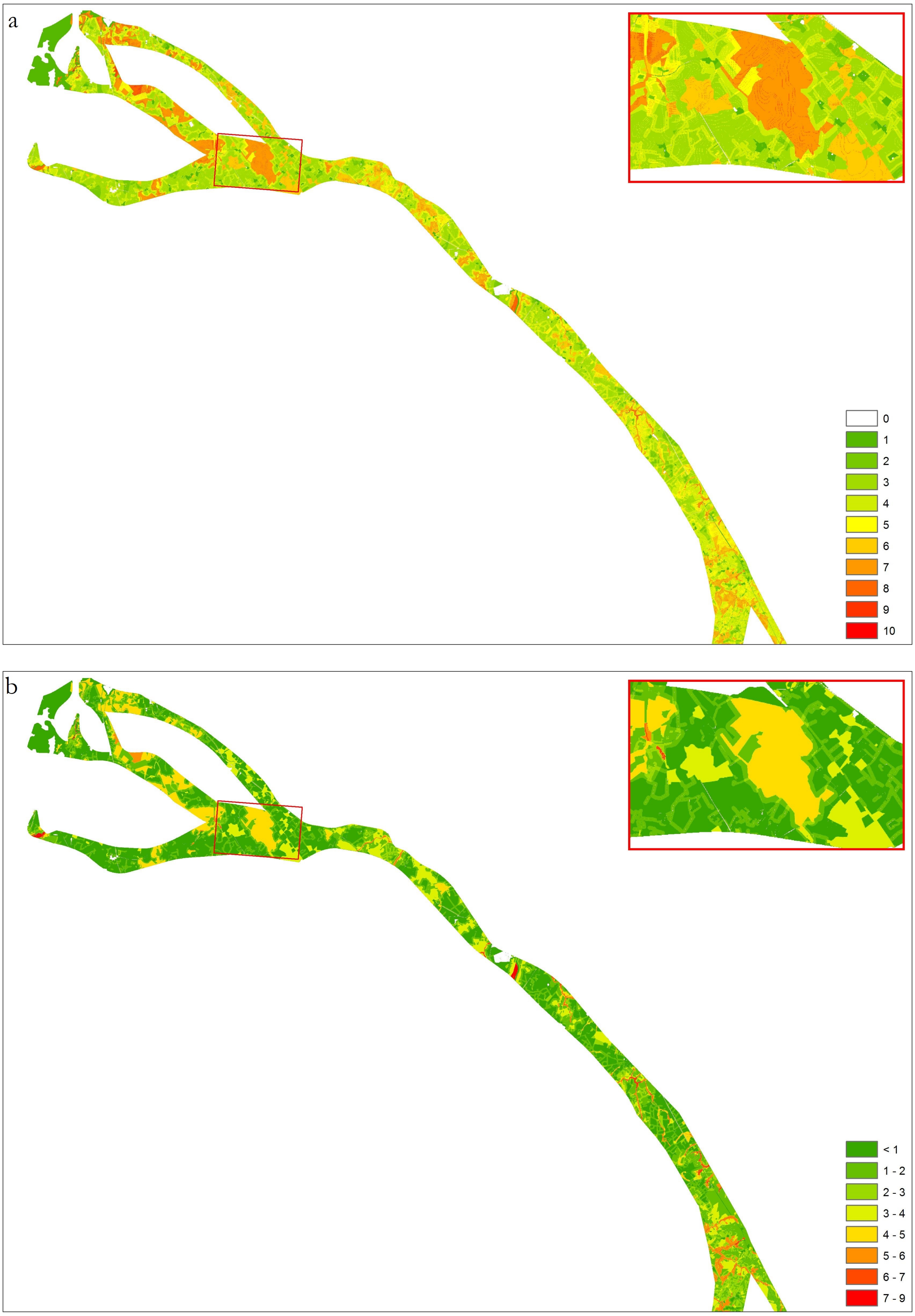



**Figure 4.3:** Indicator for preliminary environmental studies: ES presence (4.3a); presence and abundance (4.3b)

### 4.4.2 ES loss in the detailed EIA

The comparison of implementation options can be made by comparing ES loss induced by each route option for the infrastructure, highlighting the one that minimize the total loss. To perform this analysis, the characterization of the spatial extent of impacts on ES has to be added: direct impacts and additional indirect impacts on landscape connectivity threatening the deliverance of some ES at a larger scale. Impact characterization, biophysical data or ecological functions, and economic values used to assess the different route options are described in the Chapter 3 in terms of annual flows. In order to consider long term ES losses per route option, we apply the rules described in the subsection 3.2.2, for a 55 year period, that is 5 years of construction and 50 years of operation. The annual values of ES loss per route option (expressed in real terms), then evolute in function of first, an increase relative prices (represented by income elasticities of WTP and the real income rate per capita), and second, the public discount rate defined in the Robien Instruction (2005).

The long term ES losses per route option and per ES category are presented in Table 4.5. The total monetary loss for the different route options yield to the same trade-off among options than annual values. Trade-offs regarding ES loss are just strengthened. Results still show a large predominance of regulation services, with a loss ranging from 4 to 6 M€ for 55 years. Route 1.3 remains the least impacting route in terms of aggregated values over 55 years. The major increase between annual and long term values are observed for recreation services without changing the ranking among options. Route 2.1 remains also the least impacting route and the major increase is also observed for recreation services. The repartition of the loss per service for zone 1 is presented in figure 4.4. We can underline as in the previous chapter the predominance of the global climate service, which is highly valued in comparison to other services. Here the air quality regulation is also important given the presence of peri-urban forests in the area.

EIAs are constructed so as to answer the question: "what are the most significant likely environmental impacts of the project?". It is a qualitative measure based on expert knowledge. This ES approach would adapt this question in order to ask: "what are the most important ecosystem services being provided and impacted in the area?". By comparing the results presented in the table 4.5 to the environmental study conducted in the area, we can raise several coherence and incoherence (we present the results only for zone 1).

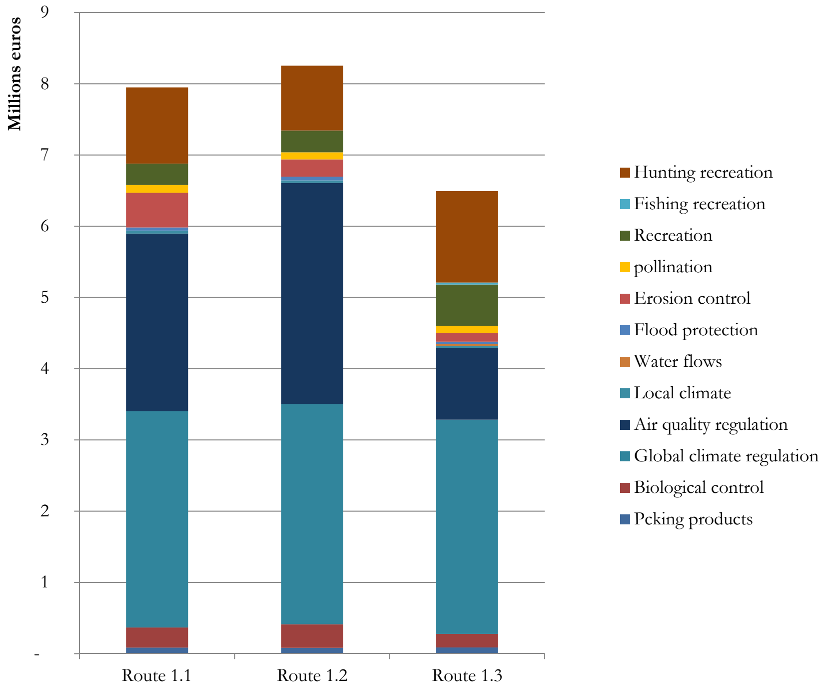



**Figure 4.4:** Repartition per of the loss per ES type for Zone 1

**Table 4.5:** Ecosystem service long term loss (55 years) per route option in two Zones

| | Zone 1 | | | Zone 2 | |
|---|---|---|---|---|---|
| In euros for the year 2010 | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
| Provision (55 years) | 85, 129 | 81, 955 | 87, 393 | 202, 629 | 276, 816 |
| Provision (annual loss) | 3, 072 | 2, 958 | 3, 154 | 7, 312 | 9, 990 |
| Regulation (55 years) | 6 493, 174 | 6 955, 666 | 4 514, 617 | 5 515 801,35 | 6 449, 780 |
| Regulation (annual loss) | 232, 394 | 249, 092 | 161, 037 | 195, 026 | 229, 490 |
| Recreation (55 years) | 1 370, 821 | 1 215, 813 | 1 891, 155 | 1 282, 773 | 1 257, 366 |
| Recreation (annual loss) | 46, 437 | 41, 186 | 64, 064 | 43, 454 | 42, 594 |
| Total loss (55 years) | 7 949, 124 | 8 253, 434 | 6 493, 165 | 7 001, 203 | 7 983, 962 |
| Total loss (annual loss) | 281, 903 | 293, 235 | 228, 254 | 245, 793 | 282, 073 |
| Loss per km (55 years) | 393, 554 | 412, 372 | 290, 651 | 339, 947 | 352, 825 |
| Loss per km (annual loss) | 13, 957 | 14, 651 | 10, 217 | 11, 935 | 12, 465 |

*Shaded cells highlight the least impacting routes*

In respect to the global consistency, the EIA and the ES based approach conclude on the same result, the route 1.3 is the least impacting route. Both approaches also point an important impact on natural areas on the route 1.2. However, some incoherence on specific points can also be noted.

First a low impact natural areas was detected for route 1.1 when the ES approach shows an important impact on ES (and particularly on regulating services). Concerning the impacts on recreation and tourism, the route 1.2 was designated as suffering of a quite strong impact, when a low impact has been identified for route 1.1. When we assess the recreation service the impact between the two routes is not as clearly discriminant. A quite low impact on agriculture was identified for routes 1.2, however the ES approach show a strong impact on pollination, local climate regulation, erosion control, and biological control for this option. Finally impacts on landscape connectivity for wild fauna were identified as low for the route 1.1 and strong for the route 1.2 and quite strong for the route 1.3. More broadly, The ES approach show that the overall impact on ES related to landscape connectivity (water flow and flood regulation, pollination, hunting, fishing, and general outdoor recreation) can also be important in route 1.1.

Hence, the ES based approach support some arguments by giving quantitative information (as the identification of route 1.3 as the least impacting route), and challenges others. It puts environmental issues into perspective within a larger territory, which involves more stakeholders, and is less confined to the project area. In terms of added information to the general public, to stakeholders and decision makers, it seems relevant to communicate such information on differences between options.

Figure 4.5 provides a map of direct and indirect ES loss (over a 55 period) in Zone 1. This type of maps may be provided in order to help the location of the best options for mitigating measures. We tested the possibility to compare the cost of one mitigating measure with the cost of ES loss. The mitigating measure analyzed here is a wildlife passageway for large game fauna to implement in the red squared zone of the Figure 4.5. To do so, we compute the difference between the loss before and after the construction of the wildlife passageway. The optimal position of the corridor (in terms of habitats and migration) in the red zone is determined with Optiflux. The service is then valued (as in the Chapter 3) as a loss of huntable territory. The difference in hunting recreation service delivery, between the situation without corridor and with corridor, is estimated around 17 thousand euros 2010 for one year and around 507 thousand euros 2010 for 55 years. The cost of the wildlife passageway is estimated at 650 thousand Euros. The construction of the passageway is not economically justified over a 55 years period if we consider only this service, however the conclusion is reversed if benefits are evaluated from t = 0 to infinity (1.16 M€). It appears cost efficient after the 81st year. It can be pointed that

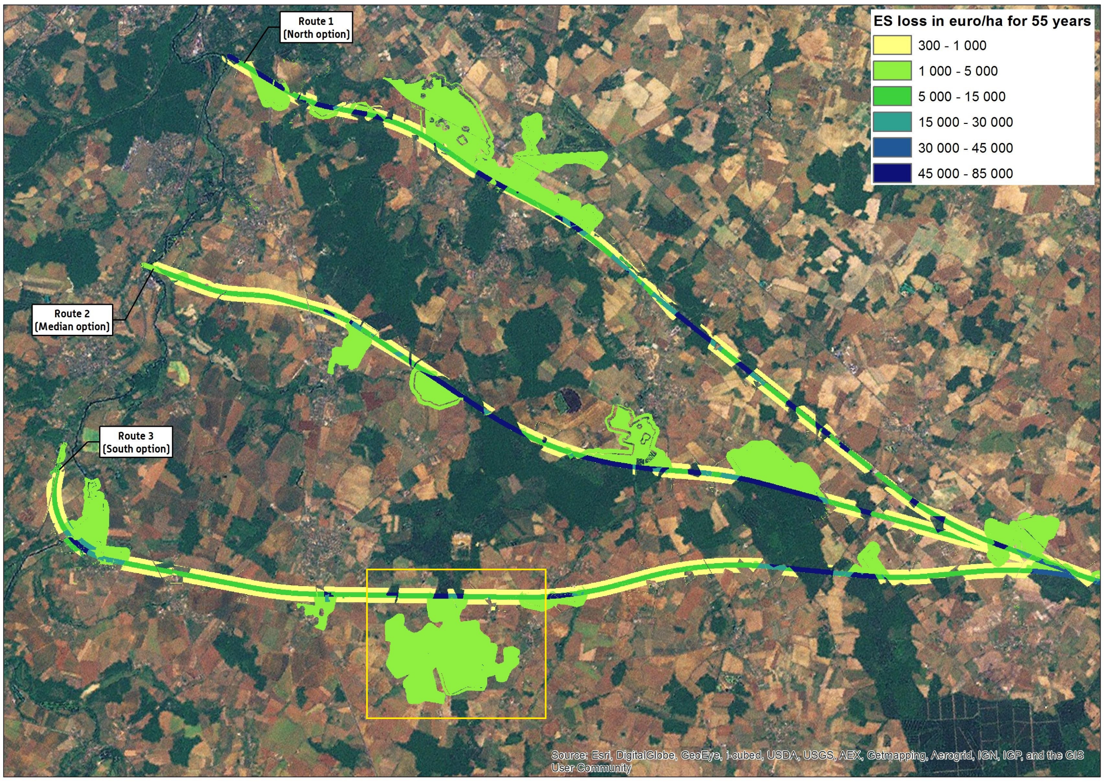



**Figure 4.5:** ES loss for 55 years for Zone 1

the only service considered (and valued) here as related to wild game fauna is the hunting recreation service, however other services can be considered as habitat services or gene-pool protection.

### 4.4.3 ES loss related to the final option for CBA

To conclude this case study, the NPV test is applied to the final option chose for the infrastructure cost benefit analysis (CBA). The CBA is computed for the case of 5 years of a construction period and 50 year life span, assuming that the infrastructure will be operating by year 2020 (this is the assumption made for the real project evaluation). The loss of ES and the construction costs start from the first year of construction, whereas benefits and operating costs start from the moment where the infrastructure is in operation.

With the assumptions explained above (Subsubsection 4.3.2), the total loss of ES is valued at 44 M€ for the year 2010. Regarding the project overall NPV, the project was already not profitable by having a negative NPV (-236 M€) and an internal rate of return lower than the social discount rate (3.34%). Accounting for ES loss results in the project being even less profitable, increasing the project negative NPV by 19%. However, it is interesting to have a look at the weight of ES loss in the others actors CBA (third parties, government, users) as presented in Figure 4.6. Accounting for ES loss diminishes the third parties balance by 42% which is not negligible. We can also see that the ES loss is in the same order or greater than the others externalities accounted for in the current CBA.

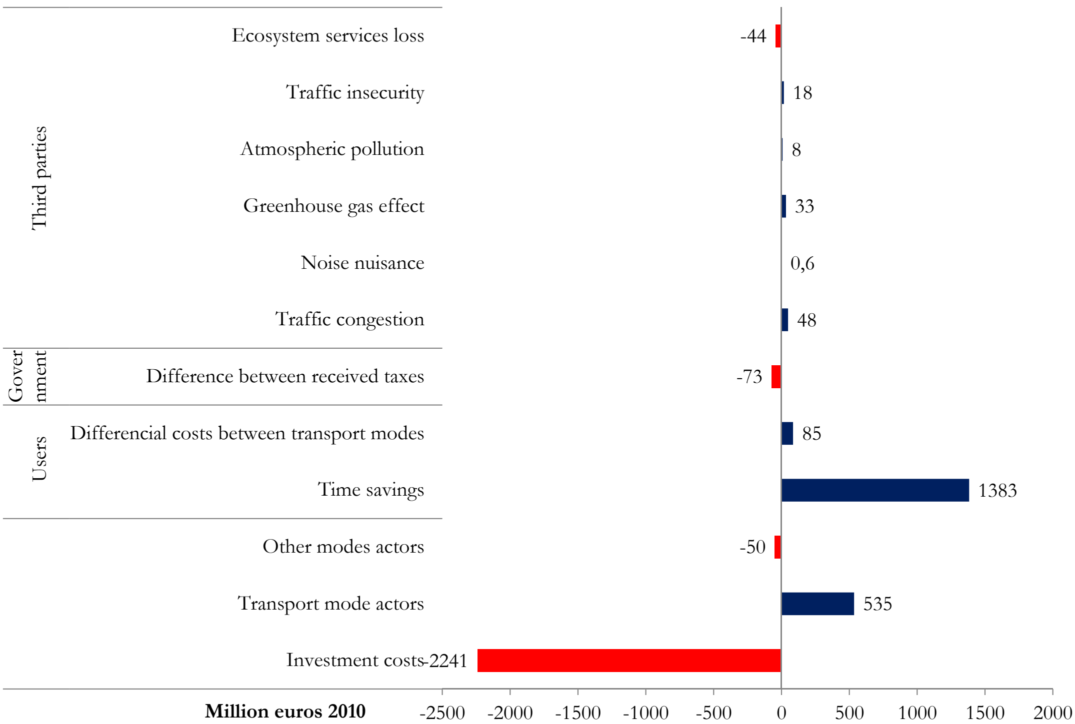



**Figure 4.6:** CBA including ecosystem services loss (55 years)

The results presented here are obtained under the assumption that the stock of carbon is released in the atmosphere during the first year of construction (Option 1 in Chapter 2). That is we do not account for the storage service (option 2 in the Chapter 2). Under this assumption, the general ES loss is of the same order of magnitude than the other nuisances, particularly traffic congestion and atmospheric pollution. Under the assumption of a storage service (option 2 in chapter 2) for the global climate regulation service loss, the loss of ecosystem services is estimated at 66 M€ for year 2010, and carbon takes a prominent role in the ES loss.

## 4.5 Discussion

The case study results supports the vision that the ES inclusion in transport infrastructure evaluation can bring additional relevant information for choices to be made at every stage of the project. The analysis provided here is constructed so as to be adaptable to the data available at each stage of the project. This inclusion supports the point that ecosystems, and their capacity to provide sustained ES benefiting to society, may constitute indications for management planning.

First, the presence/abundance combined indicator can be useful at the very preliminary stages of the project to highlight ES "hotspots" areas in terms of service supply, demand (location of beneficiaries). It can thus help in the design of first avoidance measures (in order to avoid these areas with high priority). Further, at the stage of comparison of implementation option, the analysis shows the order of magnitude of ES loss per route option providing additional information to select the least impacting route in terms of environmental and socio-economic outcomes. Ecosystem services can make the arguments less binary (environment versus economic development), emphasizing wider economic benefits of certain habitats and land cover types, *e.g.* wetlands. ES based approach may also enable to locate areas where the loss is the most important, enabling to identify the more appropriate location of mitigation measures. Mitigation measures can be hereafter tested relying on a cost-effectiveness analysis. Finally, even though the computation of ES loss for the final selected route may be considered as too dependent on assumptions to be integrated to the general CBA, it can be used to provide indicative valuations that may help improve guidance for decision-makers and the quality of general public information (as suggested by the Quinet et al, 2013 report).

In this section we present a sensitivity analysis of the total ES loss estimated according to different changes in assumptions previously made, and a discussion on the limitations and possibilities to include ES in evaluation of transport infrastructure projects.

### 4.5.1 Sensitivity analysis

In order to test the sensitivity of the ES loss estimated for the final selected route, the ES loss NPV is re-computed with different values of the following:

- *Project life span:* development benefits will not last forever, since the infrastructure schemes have a fixed life span. To be applicable to the infrastructure CBA, we calculated the loss of ES over the 55-year period, however the loss is likely to be incurred for a period far superior to this one. When the ES loss is evaluated over an infinite horizon, it results in a loss of 69.2 M€. The loss is then largely changed depending on the time-horizon under consideration.

- *Discount rate*: Using a lower discount rate (3%) in our computation, decreasing at the same pace than the public discount rate, result in an increase in the ES loss from 44 M€ to 46.4 M€. Using a 4% constant discount rate results in a loss of 38.9 M€.

- *Land-use/land-cover typology*: When the ES loss is estimated with a less precise scale typology, it results in an underestimation of the loss (41 M€). This can be mainly explained by the fact that the CLC typology overlooks spatial details that the Corine Biotope typology can detect. More precisely, the CLC typology cannot capture the same ES presence as the Corine Biotope typology can.

- *Relative prices:* We consider different relative prices, depending on four levels of income elasticity of WTP applied to all services (0, 0.3, 0.5 and 0.7). With these elasticities values of the ES losses are equal to 41.5, 43.6, 45.2, and 46.9 M€ respectively.

A summary of the ES loss according the different assumptions presented is illustrated in Figure 4.7. The 0 level represent the estimates we made in this study, other estimates are made by considering the other assumptions in terms of discount rate, life span, or relative prices evolution. For the most part of the assumptions made the NPV variation is quite negligible. However, accounting for the loss in a very long-term period changes in a consequent way the assessment of the loss. This necessarily emphasize the issue of the period to consider when accounting for ES loss.

### 4.5.2 The inclusion of ES in debate

The Quinet (2013) report argues that ES loss valuations are unreliable as related methodologies stand currently, and suggests a better clarification of environmental measures costs (avoidance, mitigation, compensation) as an alternative to integrate environmental costs in the project CBA. However, we believe that this latter approach does not bring the same information. First, because some services are totally ignored in the process and there is no effort made to avoid, reduce or mitigate the impact on such services. Specifically, it is the case for the service presented in the Table 4.3 not taken into account by the current EIA framework. Consequently, the loss incurred by local third parties under consideration in the ES approach on the one hand (*e.g.* farmers, urban and peri-urban populations),

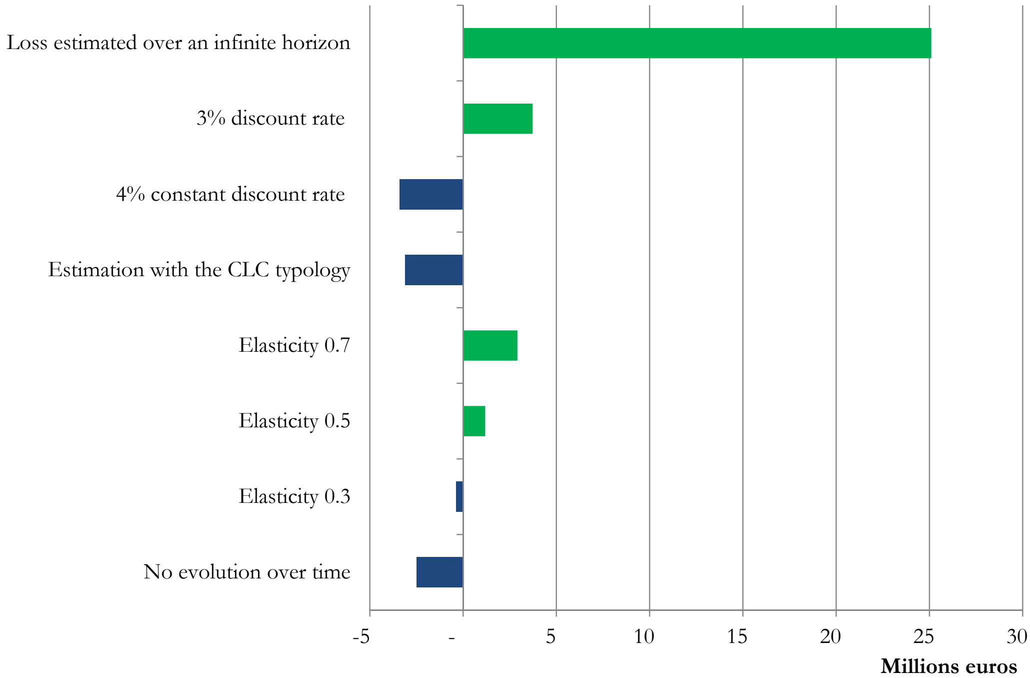



**Figure 4.7:** Difference in ES loss estimate according to different assumptions

and the global loss incurred at the global scale (global climate regulation through carbon storage), will never be taken into account. Moreover, other services are only partly and potentially accounted for through mitigation measures (as it is the case for recreation services) and compensation measures (regulation of water flows, flood prevention, local climate and air quality regulation, and provisioning services). These measures are moreover not tackled in a "service restoration" point of view, then the service can be lost in an area if the measure is not implemented in the same location. We can then presume that the real ES loss incurred will not be totally reflected by the measures costs. Ideally, one would need to assess only residual impacts that would account for the effect of mitigating and compensation measures on the ES supplies, demands and values. However this requires to assess the measures effectiveness, knowing their costs, and above all to precisely locate the measures. Without these data, the assessment is made in terms of gross impacts, that can be precised further when the data are made available. Moreover, the ES loss analysis can contribute to the development of more accurate avoidance measures, which in turn may change the cost of such measure.

Criticism to general CBA need also to be pointed. Beyond the imprecision it holds, the use of CBA tends to give more weight to the preferences of the richest agents, since they have generally higher willingness's to pay (as income elasticity for normal goods makes

WTP increase with income). Social choice theory considers the possibility to attribute unequal weights to gains and losses of different groups within the society (Fleurbaey et al, 2013; Hanley and Spash, 1993). This consideration might be important for ES analysis. The calibration of weights needs further study in order to take accurately account for individual preferences, and adjust social welfare function according to inequality aversion. So far, such weighting applied to practical CBA is unusual in the OECD countries. We abstract from such considerations in the present contribution.

The frontiers of ES scarcity are also difficult to define. Many issues raises and relates either on the supply side or on the demand side of ES (Baumgärtner et al, 2006). On the demand side the distinction is based on whether the satisfaction of needs is considered "necessary for life" or not. Necessary for life meaning that the system sustains human life and reproduction; for example for eating, drinking, sleeping, shelter, heating and basic health care. On the supply side, ecological systems are governed by different scale effects, discontinuities, thresholds, minimum viable population sizes, limited resilience, and irreversibility for some of them. Thus, when scarcity concerns a limited or non-substitutable means for the satisfaction of basic needs, and cannot be obtained by additional production, one may speak of absolute scarcity. Notions of scarcity are moreover strongly depend on time and spatial scales. Technical progress may result in new technologies over time, which then yields production of substitutes for previously absolutely scarce goods, which would in turn become scarce only in a relative sense. In the same way, a good absolutely scarce on a given spatial scale, may be relatively scarce on a larger spatial scale (or conversely).

These issues are related to the discussion on weak and strong sustainability initiated by Dasgupta and Mäler (2000), discussion that may depend on one's own perception. Proposal of weak sustainability support the fact that the market economy growth is maintained by different capital stocks used to produce goods and services (natural, manufactured, and human capital). This implicitly assumes that all capital stocks are substitutable among others. Proposal of strong sustainability assumes non-substitutability among human made capital and natural capital, considering that capitals stocks are complements. It thus calls for a separate maintenance of the natural capital stock and of the human-made capital stocks. If this latter proposal applies, humans cannot degrade or deplete any element of ecosystem structure faster than it can restore itself without eventually crossing some threshold. Valuation implies either weak sustainability or a safe distance from ecological thresholds. The fact that monetary values are exchange values implies some degree of substitutability. This is obviously not ethically neutral. However, we deal here with ecosystems that are not protected by any conservation status, and thus not directly considered by environmental measures. Our contribution is thought in order to improve existing instruments for projects assessments.

Finally, when it comes to ES valuation, there is always the risk to misguide policy-makers because valuation is an uncertain process. Moreover, interests groups may want to use the estimates based on prices to justify their own interests. Therefore, valuing the flows and stocks of nature needs to be scientifically credible with a clear objective. However, other effects are valued with non-market valuation methods and are integrated in the project process even if they hold as well many imprecision. This is the case for time savings playing an (excessive) major role in projects evaluation, as it is the case in our case study. This role could be reduced due to the increased comfort that can benefit travelers, or the fact that it is now easier to work in a train (possibility to connect the computers etc.). It can therefore be expected that every minute saved may be less benefiting in comparison with travels in 2001.

## 4.6 Conclusion

Although methodologies for the classification, quantification and valuation of ES are developing rapidly, most studies are restricted to general evaluations at regional or more global spatial scales and are rarely directly integrated into decision-making processes (Laurans et al, 2013). Yet, the key challenge of sustainable development lies in the need to make changes in all the development policies that are critical for the ES provision (Kumar et al, 2013). Thus it can be seen as necessary to integrate ES into conventional development projects as a priority, at every stages of their process, from design to implementation. This apply to transport infrastructure projects which are acknowledged major drivers of fragmentation in Europe, and one of the major drivers of habitat loss (Hicks et al, 2002). We have shown that the ES based approach can be useful and applicable from preliminary environmental studies to final route assessment. We demonstrate this feature by different metrics in terms of presence, impacts, supply/demand and values.

More precisely, the valuation of ES loss could be useful for decision making during the choice of implementation option. Currently, after the regulatory avoidance of sites designated for nature conservation, decisions are often made on the basis of environmental vulnerability, the technical aspects of the infrastructure construction, security, and short-term economic criteria based on technical costs (clearing, elevation, house protection). An approach that integrates ES loss could refine this decision-making process, and should allow convey the importance areas (affected by the project) of less remarkable biodiversity, that do not contain the emblematic or protected habitats and species that currently do not provide a basis for avoiding, mitigating or compensating effects. Assessing ES loss could allow for a broader identification of socio-economically significant environmental impacts (Landsberg et al, 2011) and could thus improve efforts to inform decisions among alternative projects for land-use planning decisions (Geneletti, 2011). All of this in an administrative context in which project managers and developers are more and more constrained by requirements to more fully integrate the larger-scale environmental dimensions of their projects, without having at their disposition sufficiently clear and applicable tools to do so (Broekx et al, 2013; Geneletti, 2013). Identifying the loss of ES associated with land development is thus a major current challenge to the improvement of terrestrial transport infrastructure and environmental planning (Geneletti, 2013; Kumar et al, 2013).

This analysis can be seen as a first real attempt of ES inclusion in the evaluation instruments of infrastructure projects. Given the state of the art on this subject, it deals with major issues rising when the inclusion is made, which in turn bring to other issues, and reveals improvements/refinement to be made. Further improvements will be, first, to integrate other services (in particular those related to water). Precisions of tradeoffs among ES can also be questioned, and can be further assessed by applying appropriate weights to services after having identified priorities with principal stakeholders in the area. Furthermore, additional research can study the characterization of residual impacts after the implementation of mitigation and compensation measures. This type of analysis can be done on the basis of the methodology provided here, applied to the final selected route, and at a more advanced stage of the project. Finally, we have seen the contribution that an ES approach can bring to better identify accurate avoidance measures, or to locate and assess the cost-effectiveness of mitigating measures; an interesting research avenue would be to study its potential articulation with compensation measures. The assessment of the ES can potentially contribute to the definition such measures, through a service-service approach for compensation (as the services are assessed also in physical terms), and the loss (or gains) can be valued in economic terms.

# References


Bagstad, K.J., Johnson, G.W., Voigt, B., Villa, F., 2013. Spatial dynamics of ecosystem service flows: A comprehensive approach to quantifying actual services. Ecosystem Services 4, 117-125.

Baker, J., Sheate, W.R., Phillips, P., Eales, R., 2013. Ecosystem services in environmental assessment - Help or hindrance? Environmental Impact Assessment Review 40.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2011. Economic analysis for ecosystem-service assessment. Environmental and Resource Economics 48, 177-218.

Baumgärtner, S., Becker, C., Faber, M., Manstetten, R., 2006. Relative and absolute scarcity of nature. Assessing the roles of economics and ecology for biodiversity conservation. Ecological Economics 59, 487-498.

Boiteux (Rapport du groupe d'experts présidé par), 2001. Transport : choix des investissements et coûts des nuisances. Commissariat Général du Plan, Paris.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators 21, 17-29.

CBD, 2012. Nagoya principle, (Online) Available from: `http://www.cbd.int/development/meetings/egmbped/nagoya-declaration-en.pdf`.

Chevassus-au-Louis, B., Salles, J.M., Pujol, J.L., Bielsa, S., Martin, G., Richard, D., 2009. Approche économique de la biodiversité et des services liés aux écosystèmes: contribution à la décision publique. La Documentation Française, Centre d'Analyse Stratégique, Paris. (Online) Available from: `http://www.ladocumentationfrancaise.fr/var/storage/rapports-publics/094000203/0000.pdfS`.

Chichilnisky, G., 1997. What is sustainable development. Land Economics 73, 467-491.

Dasgupta, P., Mäler, K.G., 2000. Net national product, wealth and social well-being. Environment and Development Economics 5, 69-93.

European Commission, 1997. Compendium of European planning systems, in: Office for Official Publications of the European Communities. Regional Development Studies (Ed.), Luxembourg.

Fisher, B., Turner, R.K., Burgess, N.D., Swetnam, R.D., Green, J., 2011. Measuring, modeling and mapping ecosystem services in the Eastern Arc Mountains of Tanzania. Prog Phys Geogr 35, 595-611.

Fleurbaey, M., Luchini, S., Schokkaert , E., Van de Voorde, C., 2012. Évaluation des politiques de santé : pour une prise en compte équitable des intérets des populations. Économie et statistique 455-456, 12-36.

Frederick, S., Loewenstein, G., O´ Donoghue, T., 2002. Time discounting and time preference : a critical review. Journal of Economic Literature 40, 351-401.

Geneletti, D., 2011. Reasons and options for integrating ecosystem services in strategic environmental assessment of spatial planning. International Journal of Biodiversity Science, Ecosystem Services & Management 7, 143-149.

Geneletti, D., 2013. Assessing the impact of alternative land-use zoning policies on future ecosystem services. Environmental Impact Assessment Review 40, 25-35.

Geneletti, D., 2013. Assessing the impact of alternative land-use zoning policies on future ecosystem services. Environmental Impact Assessment Review 40, 25-35.

Gollier, C., 2002. Discounting an uncertain future. Journal of Public Economics 85, 149-166.

Gollier, C., 2010. Ecological discounting. Journal of Economic Theory 145, 812-829.

Gowdy, J., Howarth Richard, B., Tisdell, C., 2010. Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity, in: Kumar, P. (Ed.), TEEB (2010), The Economics of Ecosystems and Biodiversity: Ecological and Economics Foundations. Earthscan, London and Washington.

Haines-Young, R., 2011. Exploring ecosystem service issues across diverse knowledge domains using Bayesian Belief Networks. Prog Phys Geogr 35, 681-699.

Hanley, N., Spash, C.L., 1993. Cost benefit analysis and the environment. Edward Elgar Publishing Limited, Cheltenham.

Hepburn, C., 2007. Valuing the far-off future: discounting and its alternatives, in: Atkinson, G., Dietz, S., Neumayer, E. (Eds.), Handbook of Sustainable Development. Edward Elgar, Cheltenham, UK, pp. 109-124.

Hicks, C., Peymen, J., 2002. Habitat Fragmentation due to Existing Transportation Infrastructure, in: Trocmé, M., Cahill, S., De Vries, J.G., Farrall, H., Folkeson, L., Fry, G., Hicks, C., Peymen, J. (Eds.), COST 341 - Habitat Fragmentation due to transportation infrastructure: The European Review. Office for Official Publications of the European Communities, Luxembourg.pp. 73-113.

Hoel, M., Sterner, T., 2007. Discounting and relative prices. Climatic Change, 265-280.

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

IEMA, 2012. Considering ecosystem services in EIA, Imagine Durban. Long term plan. (Online) Available from: `http://www.imaginedurban.org/Pages/Longtermplan.aspx2012.`

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

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E., 2011. Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, Oxford, UK.

Kriström, B., Riera, P., 1996. Is the income elasticity of environmental improvements less than one? . Environmental and Resource Economics 7, 45-55.

Krutilla, J.V., Fisher, A.C., 1975, 1985. The Economics of Natural Environments. Johns Hopkins Press, Baltimore, MD.

Kumar, P., Esen, S.E., Yashiro, M., 2013. Linking ecosystem services to strategic environmental assessment in development policies. Environmental Impact Assessment Review 40, 75-81.

Landsberg, F., Ozment, S., Stickler, M., Henninger, N., Treweek, O., Venn, J., Mock, G., 2011. Ecosystem Services Review for Impact Assessment: Introduction and Guide to Scoping, WRI Working Paper. World Resources Institute, Washington DC.

Laurans, Y., Rankovic, A., Billé, R., Pirard, R., Mermet, L., 2013. Use of ecosystem services economic valuation for decision making: Questioning a literature blindspot. Journal of Environmental Management 119, 208-219.

Lebègue, D., Baumstark, L., Hirtzman, P., 2005. Révision du taux d´ actualisation des investissements publics. La Documentation Française, Paris.

Mascarenhas, A., Ramos, T., Haase, D., Santos, R., 2014. Integration of ecosystem services in spatial planning: a survey on regional planners' views. Landscape Ecology, in press. DOI: `10.1007/s10980-014-0012-4`

Millenium Ecosystem Assessment, 2005. Ecosystem and well being : A framework for assessment. Island Press Blurb, Washington DC.

Nelson, E., Bhagabati, N., Ennaanay, D., Lonsdorf, E., Pennington, D., Sharma, M., 2013. Modeling Terrestrial Ecosystem Services, in: S.A., L. (Ed.), Encyclopedia of Biodiversity, second edition. MA: Academic Press, Waltham, pp. 347-361.

OECD., 2010. Strategic environmental in assessment ecosystem services. , (Online) available from: `http://www.oecd.org/dataoecd/24/54/41882953.pdf2010`

Oxera, 2002. A social time preference rate for use in long-term discounting, in: The Office of the Deputy Prime Minister (Ed.). Department for Transport and Department for the Environment, Food and Rural

Affairs, , London.

Pearce, D., Groom, B., Hepburn, C., Koundoury, P., 2003. Valuing the future. Recent advances in social discounting. World Econonomics 4, 121-141.

Pearce, D.W., 1980. The Social Incidence of Environmental Costs and Benefits, in: O´ Riordan, T., Turner, R.K. (Eds.), Progress in Resource Management and Environmental Planning. John Wiley & Sons Ltd., London.

Pearce , D.W., 2006. Framework for assessing the distribution of environmental quality, in: Serret, I., Johnstone, N. (Eds.), The distributional effect of environmental policy. OECD, Cheltenham.

Presnall, C., López-Hoffman, L., Miller, M.L., Adding ecosystem services to environmental impact analyses: More sequins on a "bloated Elvis" or rockin' idea?. Ecological Economics. In press. DOI: 10.1016/j.ecolecon.2014.02.001

Quinet, A., Baumstark, L., Célestin-Urbain, J., Pouliquen, H., Auverlot, D., Raynard, C., 2009. La valeur tutélaire du carbone. Centre d'analyse Stratégique. La documentation française, Paris.
Quinet, E., Baumstark, L., Bonnet, J., Croq, A., Ducos, G., Meunier, D., Rigard-Cerison, A., Roquigny, Q., 2013. Evaluation socioéconomique des investissements publics. Rapports et documents. Commissariat Général à la Stratégie et à la Prospective. La documentation française, Paris. (online) available from: http://www.ladocumentationfrancaise.fr/rapports-publics/134000626.

Robien, G., 2005. Instruction Cadre relative aux méthodes d'évaluation économique des grands projets d'infrastructures de transport. Ministère de l'Equipement, des Transports, du Logement, du Tourisme et de la Mer.

Ruckelshaus, M., McKenzie, E., Tallis, H., Guerry, A., Daily, G., Kareiva, P., Polasky, S., Ricketts, T., Bhagabati, N., Wood, S.A., Bernhardt, J., Notes from the field: Lessons learned from using ecosystem service approaches to inform real-world decisions. Ecological Economics, in press. http://dx.doi.org/10.1016/j.ecolecon.2013.07.009

Settle, C., Shogren, J., 2004. Hyperbolic discounting and time inconsistency in a native exotic species conflict. Resource and Energy Economics 26, 255-274.

Sterner, T., Persson, M., 2008. An Even Sterner Review : Introducing Relative Prices into the Discounting Debate. Review of Environmental Economics and Policy 2, 61-76.

TEEB, 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. In: Kumar, P. (Ed.), Earthscan, London, Washington.

Weitzman, M.L., 1998. Why the far distant future should be discounted at its lowest possible rate Journal of Environmental Economics and Managment 36, 201-208.

Weitzman, M.L., 2001. Gamma discounting. American Economic Review 91, 260-271.

WRI, 2011. Ecosystem services review for impact assessment., (Online) Available from: http://www.wri.org/publication/ecosystem-services-review-for-impactassessment2011.

# General Conclusion

This thesis proposes methods that may improve the impacts accounting of transport infrastructures on ecosystems and biodiversity. We have deepened the methods for the consideration of ecosystem services in projects evaluation.

## This thesis is placed in the cost-benefit analysis context

This work has relied on economic calculation to identify and quantify, in the cost-benefit analysis context, the (negative) effects related to non-market loss of ecosystem services caused by transport infrastructure projects. This should allow providing decision makers with a more complete outlook of the costs involved by the different operations and options between which they have to choose. The price system cannot contribute to public prosperity insofar costs for decision makers actually represent costs incurred by the community. These costs should include the externalities induced by transport infrastructure projects. The use of non-market valuation is therefore necessary in order to not consider these externalities costs to be zero.

This work was carried out in a context of increasing concern for the impact of our lifestyles on biodiversity and ecosystems, including our transport modes. It is necessary to design in better ways the infrastructure implementation through an effective consideration of biodiversity issues in project development, construction, maintenance and operation of transport infrastructures. It is for this reason and in this context that this work was conducted. Consideration of ecosystem services in infrastructure projects evaluation can indeed help to meet these objectives by (i) an extension of the number and nature of environmental externalities taken into account in the cost-benefit analysis of infrastructures, (ii) the communication on systems dependence to the natural environment, and its accounting in the choice of the options for infrastructures.

A multidisciplinary approach was necessary, including economics, ecology, and project evaluation. I have considered this approach as essential, while being aware of the risk associated with this choice, because I was not a specialist of all the involved disciplines. Considering the state-of-the-art of the consideration of ecosystem services in projects, the first objective was to better understand the issues raised by integration, in theoretical and practical ways. Furthermore, we aimed to propose a methodology relative to the identification and to the characterization of infrastructure impacts on ecosystem services supply by ecosystems while taking into account their biogeographical contexts and their functional aspects. Finally, we aimed to spatially value the loss of multiple services induced by implementation options and to explore the usefulness of spatially representing this loss. To make this method usable, we sought to determine its contribution at different stages of the project (preliminary studies, detailed environmental impact assessment, cost benefit analysis and public inquiries). We also ensured that the resulting tool is reproducible, adapted to existing techniques and acceptable by project stakeholders (policy makers, project designers, public administration, and general public). This has motivated the writing of a methodological guide.

## The thesis proposed methodologies to consider ecosystem services in the evaluation of infrastructure projects

The implementation of linear transport infrastructure affects landscape properties. Hence, the spatial aspect of ecosystem services is of particular importance in the estimation of the loss associated with implementation options. We have attempted to identify the aspects to be spatially taken into account for services, and have developed a methodological framework for a systematic approach in the infrastructure case. The methodological framework was applied to a high-speed rail project but is adaptable to highway projects.

The main two elements that cause spatial variation in services loss are the initial conditions in the studied area, and the extent of the infrastructure impact. Regarding initial conditions, after defining the services delivered by ecosystems in the studied area and determining those impacted by the infrastructure, we have quantified the ecosystem services supply in biophysical terms. For some services we had local data or data from ecosystems close to the studied conditions. Supply conditions of service have been identified in this case using spatial assumptions based on Geographical Information Systems (GIS). For other services, we have used models of ecological production for simulating supply.

For recreation services, the spatial effects were taken into account through variables such as distance from an urban area or number of substitutes. For the infrastructure impact area, we have developed an impact typology per ecosystem services and per ecosystems in order to determine the impact extent on the service. The area of impact was shown to be dependent on direct and indirect impacts of the infrastructure on functions and ecosystems connectivity. We have also proposed a way to integrate threshold effects to evaluate how they may potentially affect the outcome of ecosystem service supply. This allowed us to consider ecological network disruption or the partial conversion of an ecosystem with high functional scale in the estimation of the loss. This had never been done so far to my knowledge.

The economic valuation of loss, in monetary terms, was then performed using methods from non-market valuation, essentially with adjusted market prices, namely market costs (avoided damage costs, replacement costs or opportunity costs) or, when no data was available, value transfer. The service loss has finally been calculated and mapped for each option of infrastructure route in order to compare the loss associated with each of them.

## Take or not take into account the services in project evaluation?

The last part of this work has studied the usefulness of such integration at different stages of the evaluation of infrastructure projects, particularly at the stages of:

- Preliminary studies, in order to design the first avoidance measures (avoid these areas with highest priorities);
- Detailed environmental impact assessments, to compare implementation options, precise avoidance measures, and better locate mitigating measures that can be further tested by cost-effectiveness analysis;
- Cost-benefit analysis, to provide indicative evaluations that may help improve guidance of decision makers and the quality of general public information.

We have tested this interest by first defining the evolution over time of economic values associated with services loss (changes in prices of environmental goods). We have then suggested ways to integrate the loss of services in the current legal framework, through an indicator, and then by valuing the long-term loss given different legal aspects (market and non-market effects already integrated, their evolution in time, public discount rate, infrastructure life-span).

The sensitivity of results is presented in relation to criteria including:

- The accuracy of the land use/land cover typology, to evaluate the advantages of taking into account a precise typology for some services or specify where a less precise classification may be sufficient;

- The values margins of error , showing the sensitivity of estimates but also the fact that estimating the loss with average value, in our case, does not always change tradeoff between options;

- The infrastructure life-span, showing that the loss over the project life span (55 years) is underestimated in a consequent way compared to an estimation on a very long-term period.

We are aware that the estimates of ecosystem services impacts depend on assumptions. In the absence of a market validation, methods necessarily lack robustness. When it comes to move from principles to practice, we may feel that the data and empirical or theoretical studies are insufficient to make a reliable assessment. Waiting might seem more "reasonable", but meanwhile the nuisance inflicted on biodiversity and ecosystem services is still excluded from project assessment, and are counted as zero in the calculations. We have therefore based our work on the rationale that economies and human well-being critically depend on biodiversity and ecosystems, that the trend of erosion is not acceptable, and that a real concern about this issue has raised in the public. We believe then that it is crucial to translate current knowledge into actions that will influence development planning processes and particularly those applying to transportation. A sentence of Marcel Boiteux (2001) could also apply to our case:

> "We have to dive in (...). But, I repeat, this report is only one step, partial, in the way that will lead one day to correcting values. It cannot be overemphasized that the work presented here is inherently imperfect, periodically reviewable, but absolutely essential." Boiteux (2001).

## Research avenues

In terms of mitigation and compensation *measures*, research could be conducted to determine the residual impact on services delivery remaining after implementing these measures.

This would require that mitigation measures are identified, located and their costs determined at the stage of study design, thereby clarifying the assessment of the loss caused by the infrastructure implementation. Moreover, the link with compensation measures could also be considered. We hope that this work can be used to provide information on a “services - services” compensation approach (rather than a “value - value” compensation approach), based on quantifications proposed here, taking into account the ecosystem spatial context. This could allow identifying the cases in which positive benefits are seen after compensation measures or otherwise for which a net loss of service is still generated.

Regarding *impacts*, one could expand the impacts type considered in order to take into account the cumulative impacts with neighbouring development projects. The adaptation of this thesis to this type of impacts could be made quite simply provided that the information is available (location, commissioning year, etc.). Taking into account the induced impacts by the infrastructure (such as consecutive urbanization commissioning of the project) seems more difficult to achieve, because the information is not available at the moment when the study is applied. However, the use of data from experiences feedback of other projects might be possible. Finally, the positive impacts of an infrastructure could be considered in analyses, as for example in the case of roadsides colonized as a new habitat or serving as corridor transition for species habitat or migration function.

For *services* consideration, a more extensive assessment would obviously be desirable. This can be done by taking into account services related to water (this work is currently in progress), or provoking thought on how one can include services such as habitat maintenance or ecosystem dis-services (such as pollen causing allergies in urban areas). Any improvement in services data considered here would also be beneficial to the analysis.

*More generally*, further research could be conducted on how improving the links between the spatial assessment of the services supply, demand and values that are associated with their changes. This would allow better characterization of the relationship between a change in ecosystem service supply and the associated change in welfare. A possible option could involve the combination of mapping techniques for services supply, with techniques enabling the differentiation in willingness to pay per service unit provided by multiple ecosystems in type, size, or location.

The characterization of substitute sites to the studied ecosystems would be required, by incorporating the concept of perimeter ("market area") to validate the other sites in terms of substitutes for all services. The key criteria would be here the distance (substitutes may be the same and located at isochronous distances), and the site accessibility. Different types of ecosystems, but supplying the same service, could also be considered as a substitute. This would enable one to calculate net losses. Finally field surveys would be necessary to identify the role of distance on preferences in terms of ecosystems (due to their scarcity) or the effect of the substitute's existence. A choice experiment technique might be achievable and would improve the consideration of services loss (Liekens et al, 2010; Martin-Ortega et al, 2012, Schaafsma, 2012). The experiment could be carried out on a site and then extended on the entire area by generating a transfer function benefit (scaling-up). However, it also would lead to real challenges. It is likely that we can only determine the value of some services such as recreation, or benefits associated with non-use (existence, inheritance) and that this approach would be more difficult to implement to reveal values associated with regulating services (*e.g.* air quality, water quality, carbon sequestration and storage).

Finally, we can expect that the use of spatial econometric methods could improve the analysis. Indeed, it enables treating peculiarities of spatial data: spatial autocorrelation, which refers to the dependence between geographical observations and spatial heterogeneity, related to the spatial differentiation of variables and behaviours (Gallo, 2002). Spatial autocorrelation for a variable, enable to determine if there is a functional relationship between what happens at one point in space and what happens in a neighbouring sites. Bockstael (1996) or Anselin (2001) showed that such methods may have important applications in environmental economics, and in particular for the ecosystem services valuation.

## Références


Anselin, L., 2001. Spatial effects in econometric practice in environmental and resource economics. American Journal of Agricultural Economics 83, 705-710.

Bateman, I., Brouwer, R., Ferrini, S., Schaafsma, M., Barton, D., Dubgaard, A., Hasler, B., Hime, S., Liekens, I., Navrud, S., De Nocker, L., Sceponaviciute, R., Semeniene, D., 2011. Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe. Environmental and Resource Economics 50, 365-387.

Bockstael, N., E.,, 1996. Modeling economics and ecology: the importance of a spatial perspective. American Journal of Agricultural Economics 78, 1168-1180.

Commisariat Général du Plan, Rapport du groupe d'experts présidé par M Boiteux, 2001. Transport : choix des investissements et coûts des nuisances, Paris.

Le Gallo, J., 2002 Économétrie spatiale : l'autocorrélation spatiale dans les modèles de régression linéaire. Economie & prévision 155, 139-157.

Liekens, I., Schaafsma, M., De Nocker, L., Broekx, S., Staes, J., Aertsens, J., Brouwer, R., 2013. Developing a value function for nature development and land use policy in Flanders, Belgium. Land Use Policy 30, 549-559.

Martin-Ortega, J., Brouwer, R., Ojea, E., Berbel, J., 2012. Benefit transfer and spatial heterogeneity of preferences for water quality improvements. Journal of Environmental Management 106, 22-29.

Schaafsma, M., Brouwer, R., Rose, J., 2012. Directional heterogeneity in WTP models for environmental valuation. Ecological Economics 79, 21-31.

# Conclusion Générale

Cette thèse a permis de définir des pistes pour une prise en compte plus explicite et mieux argumentée de l'impact des projets d'infrastructures de transports terrestres sur les écosystèmes et la biodiversité. Nous avons approfondi les méthodes permettant la prise en compte des services écosystémiques dans l'évaluation de ces projets en tentant de dépasser le simple stade de la réflexion.

## Cette thèse s'est placée dans le cadre de l'analyse coût-avantage

Ce travail s'est inscrit dans le cadre du calcul économique visant à identifier et de quantifier, dans le cadre de l'analyse coût-avantage, les effets (négatifs) non-marchands relatifs à la perte de services engendrée par les projets d'infrastructure de transport. Ceci permet de fournir aux décideurs un panorama plus complet des coûts engendrés par les diverses opérations et variantes entre lesquelles ils ont à choisir. Le système des prix ne peut contribuer à la prospérité publique que dans la mesure où les coûts pour les décideurs représentent réellement les coûts supportés par la collectivité. Or, ces coûts devraient inclure les externalités induites par les projets d'infrastructures de transport. Le recours à l'évaluation des effets non-marchands est donc nécessaire pour ne pas considérer le coût de ces externalités comme nul.

Ce travail a été réalisé dans un contexte de préoccupation croissante pour l'impact de nos modes de vie sur la biodiversité et sur les écosystèmes, et notamment de nos modes de transport. Il s'agit de mieux prévoir l'implantation des infrastructures par une meilleure prise en compte des enjeux relatifs à la biodiversité dans l'élaboration des projets, la construction, l'entretien et l'exploitation des infrastructures de transports terrestres. C'est à ce titre et dans ce contexte que ce travail a été conduit. La prise en compte des services écosystémiques dans l'évaluation des projets d'infrastructures peut en effet contribuer à répondre à ces deux objectifs par (i) un élargissement des externalités prises en compte dans les bilans afin de permettre leur internalisation grâce aux méthodes développées par l'analyse économique, (ii) la communication sur la dépendance de nos systèmes à l'environnement et sa prise en compte dans les choix de tracés d'infrastructure.

Une approche pluridisciplinaire s'est imposée, incluant économie, écologie, et évaluation de projet en matière de transport. J'ai considéré cette approche comme indispensable, tout en étant consciente du risque associé à ce choix puisque je n'étais pas spécialiste de toutes les disciplines impliquées. Compte tenu de l'état de l'art de la prise en compte des services écosystémiques dans ces projets, l'objectif premier était de mieux comprendre les enjeux d'une telle intégration, tant au niveau théorique que pratique. L'objectif suivant était de proposer des pistes méthodologiques relatives à l'identification et à la caractérisation des impacts sur la fourniture des services rendus par les écosystèmes tout en prenant en compte leurs contextes biogéographiques et leurs aspects fonctionnels. Enfin, la finalité était d'évaluer la perte des différents services dans les choix d'implantation des infrastructures de transport et d'explorer l'intérêt des représentations (spatiales) de cette perte. Pour rendre cette méthode utilisable, nous avons cherché à déterminer son apport à différents stades du projet (études préliminaires, avant-projet sommaire, enquêtes publiques) dans le cadre réglementaire en vigueur pour ces projets. Nous avons également cherché à ce que cet outil soit reproductible, adapté aux techniques existantes ainsi qu'acceptable pour les acteurs des projets (décideurs, maîtres d'ouvrage, administration et public à un sens plus large). Ceci a motivé la rédaction d'un guide méthodologique.

## La thèse propose des méthodologies de prise en compte des services écosystémiques dans l'évaluation des projets d'infrastructures

Les infrastructures de transports linéaires modifient les propriétés des territoires. La dimension spatiale des services écosystémiques prend donc une importance particulière dans l'estimation de la perte subie sur l'ensemble des tracés. Nous avons donc tenté d'identifier les différents aspects à prendre en compte pour spatialiser les services, et développé un cadre méthodologique pour systématiser l'approche dans le cas des infrastructures de transport. Le cadre méthodologique a été appliqué à un projet de construction de ligne à grande vitesse mais est adaptable aux projets d'autoroutes.

Les deux principaux éléments déterminés comme faisant varier la perte de service dans l'espace sont les conditions initiales de la zone d'étude, puis l'aire d'impact de l'infrastructure sur ces services. Concernant les conditions initiales, différentes étapes ont été réalisées. Après avoir défini les services rendus présents dans l'aire d'étude et déterminé ceux impactés par l'infrastructure, nous avons réalisé la mesure de l'offre de service. Pour certains services nous disposions de données locales ou d'écosystèmes proches des conditions d'études. Les conditions d'offre de service ont été précisées par différentes hypothèses spatiales basées sur des Systèmes d'Information Géographique (SIG). Pour d'autres services, des modèles de fonction de production écologiques ont été utilisés pour simuler l'offre. Pour la récréation, des effets spatiaux ont été pris en compte à travers des variables telles que la distance par rapport à une aire urbaine ou le nombre de substituts. Pour l'aire d'impact de l'infrastructure, nous avons développé une typologie d'impact par type d'écosystèmes et par service afin de déterminer l'étendue de l'impact sur le service. L'aire d'impact a été déterminée comme dépendante des impacts directs et indirects de l'infrastructure sur les fonctions et la connectivité des écosystèmes. Nous fournissons également une manière d'intégrer un effets de seuil pour évaluer leur incidence sur la fourniture de service. Ceci nous permet de considérer des phénomènes comme la perturbation d'un réseau écologique ou l'artificialisation d'écosystèmes ayant une grande échelle fonctionnelle dans l'estimation de la perte. Ceci n'avait pas encore été réalisé à ma connaissance.

La valorisation de la perte en termes monétaire a ensuite été réalisée en utilisant des méthodes issues de l'évaluation économique non-marchande, essentiellement des prix de marché ajustés, des méthodes de valorisation par les coûts (coûts évités ou de remplacement, coût d'opportunité, coût des dommages) ou, lorsque qu'aucune donnée n'était disponible, du transfert de valeur. La perte de service a été calculée et cartographiée pour chaque option de tracé de l'infrastructure de manière à comparer les pertes de services engendrées par chacune d'elles.

## Prendre ou ne pas prendre en compte les services dans l'évaluation des projets ?

Dans la dernière partie de ce travail nous avons étudié l'intérêt de cette intégration dans le processus d'évaluation des projets d'infrastructures, notamment au niveau :
- Des études préliminaires, dans la définition des grandes mesures d'évitement à mettre en oeuvre dans les différents corridors de passage.

- Des études d'impact plus avancées au moment de la comparaison de tracés, afin de préciser les évitements, et d'être en mesure de mieux situer les mesures de réduction ou tester leur coût-efficacité.
- Du bilan socio-économique afin de communiquer sur l'importance de la perte engendrée par le projet, avec une vision large des parties prenantes impactées par le projet.
Nous avons testé cet intérêt en ayant d'abord défini l'évolution dans le temps des valeurs économiques associées à la perte des services (évolution des prix des actifs environnementaux). Nous avons ensuite proposé des pistes pour l'intégration de la perte de services dans le cadre réglementaire actuel, à travers un indicateur, puis en évaluant la perte à long terme compte tenu des variables réglementaires à prendre en compte (effets marchands et non marchands déjà intégrés, taux d'actualisation, durée de vie de l'infrastructure).

La sensibilité des résultats est présentée par rapport à certains critères, notamment :
- La précision de la typologie de couverture des terres disponibles - permettant de préciser l'avantage de prendre en compte une typologie précise pour certains services ou de préciser les cas où une typologie moins précise peut suffire ;
- Les marges d'erreur des valeurs, montrant la sensibilité des estimations mais aussi le fait qu'elles ne font pas nécessairement changer d'arbitrage entre tracés ;
- La durée de vie de l'infrastructure, montrant que la perte est largement sous-estimée lorsque l'on se restreint à cette durée de vie.

Nous sommes conscients que les estimations des impacts sur les services écosystémiques sont dépendantes des hypothèses. En l'absence de validation par un marché, les méthodes manquent nécessairement de robustesse. Quand il s'agit de passer des principes à la pratique, on peut avoir le sentiment que les données, les études empiriques ou théoriques, sont encore insuffisantes pour en faire un travail robuste. Attendre pourrait paraître plus "raisonnable", mais pendant ce temps les nuisances infligées à la biodiversité et aux services liés aux écosystèmes continuent d'être écartées des bilans. Elles sont donc comptées comme nulles dans les calculs. Nous sommes donc partis de l'hypothèse que notre bien-être et que nos économies reposent de manière critique sur la biodiversité et les écosystèmes, que la tendance d'érosion qu'ils connaissent aujourd'hui n'est pas acceptable, et qu'il existe maintenant une réelle préoccupation autour de cette question. Nous pensons donc qu'il est crucial de traduire les connaissances en actions notamment au niveau des projets de développement qui vont influencer cette tendance, comme c'est le cas des infrastructures de transport. Une phrase de Marcel Boiteux (2001) pourrait s'appliquer également à notre cas :

> “ Force est donc de se jeter à l’eau (...). Mais, répétons-le, ce rapport ne constitue qu’une étape, partielle, dans la voie qui aboutira un jour, de retouches en retouches, à des valeurs de mieux en mieux fondées. Car on ne saurait trop souligner que le travail ici présenté est intrinsèquement imparfait, périodiquement révisable, mais tout à fait indispensable.” M. Boiteux (2001).

## Recherches futures

Au niveau des *mesures* de réduction et de compensation, une réflexion pourraient être menée de manière à déterminer l’impact résiduel demeurant sur la provision de service suite à la mise en place de celles-ci. Ceci nécessiterait que les mesures de réduction soient identifiées, localisées et que leur coût soit déterminé au moment de l’étude. Mais la perte engendrée par la mise en place de l’infrastructure en serait grandement précisée. Par ailleurs, le lien que l’on pourrait faire avec les mesures de compensation pourrait également être étudié. Nous pourrions envisager que le travail réalisé puisse servir à alimenter la réflexion concernant une approche de compensation de type “services-services” (et non “valeur-valeur”) basée sur les quantifications proposées ici, prenant en compte le contexte spatial des écosystèmes. Ceci pourrait permettre d’identifier les cas où des avantages positifs sont observés après compensation ou au contraire ceux pour lesquels une perte nette de service est générée.

Pour les impacts on pourrait élargir le type d’*impacts* de manière à prendre en compte les impacts cumulés (avec d’autres projets de développement alentours). Nous ne l’avons pas fait ici, cependant l’adaptation de ce travail à ce cas pourrait être réalisée assez simplement à condition que les informations soient disponibles (localisation, année de mise en service, etc.). La prise en compte des impacts induits par l’infrastructure (comme l’urbanisation consécutive à la mise en service du projet) parait plus difficilement réalisable compte tenu des informations disponibles au moment de l’étude. Cependant, l’utilisation de données de retours d’expériences sur d’autres projets pourrait être envisageable. Enfin une réflexion sur la prise en compte d’impacts positifs de l’infrastructure pourrait être engagée, comme par exemple les bords des routes colonisés comme nouvel habitat ou servant de corridor de transition (fonction d’habitats ou de conduits de déplacement).

Pour la prise en compte de *services*, une évaluation plus exhaustive serait évidemment souhaitable. Ceci passerait donc par la prise en compte notamment des services liés à l'eau (ce travail est actuellement en cours), voire à engager une réflexion sur la prise en compte de services tel que le maintien d'habitat ou de dis-services écosystémiques (comme le pollen provoquant des allergies dans les aires urbaines). Toute amélioration des données concernant les services pris en compte ici serait également bénéfique à l'analyse.

*Plus généralement*, des recherches plus approfondies pourraient être menées sur la manière de faire le lien entre l'évaluation de l'offre de service à leur demande et aux valeurs qui sont associées à leurs changements. Ceci permettrait de mieux caractériser la relation qui existe entre un changement au niveau de la fourniture d'un service et le changement de niveau de bien-être qui en découle. Une option possible serait de combiner des études de cartographie de l'offre de service avec des techniques permettant de différencier le consentement à payer par unité de service fourni par différents écosystèmes (en types, en taille, ou en localisation).

Une réflexion sur la caractérisation des sites substituts aux écosystèmes étudiés serait alors requise. Un raisonnement intégrant des notions de périmètre ("aire de marché") pour valider les autres sites en tant que substituts pourrait être envisageable. Le critère-clé serait alors la distance (les substituts pourraient être des sites identiques et situé à des distances isochrones), et l'accessibilité de ces sites [4]. Un écosystème de type différent, mais fournissant le même service, pourrait également être considéré comme substitut. Il s'agirait alors de calculer les pertes nettes étant donné la valeur des sites substituts. Ensuite, une réflexion sur la manière de révéler la demande effective par des enquêtes de terrain serait nécessaire : identifier le rôle de la distance sur les préférences, les préférences en termes d'écosystèmes (compte tenu de leur rareté, de leur taille, etc.) ou l'effet de l'existence des substituts. Une méthode d'expérience de choix ("choice experiment") ici pourrait être réalisable et contribuerait à améliorer la prise en compte des pertes de services (Liekens et al, 2010 ; Martin-Ortega et al, 2012, Schaafsma, 2012). L'expérience pourrait être menée sur un site puis étendue sur l'ensemble de l'aire par la génération d'une fonction de transfert de bénéfice (et procéder à du "scaling-up"). Cependant, elle entrainerait aussi de réels défis. Il est par exemple possible qu'on ne puisse qu'établir la valeur de certains services comme la récréation, ou de bénéfices associés à des non-usages (existence, patrimonial), et que ce type d'approche soit plus difficile à mettre en place pour révéler les valeurs associées

[4] L'étude se rapprochant le plus de cela est l'étude de Bateman et al (2011) dans laquelle ils considèrent le site le plus proche comme substitut.

aux services de régulation par exemple (qualité de l'air, qualité de l'eau, séquestration et stockage du carbone).

On peut enfin s'attendre à ce que l'utilisation de méthodes d'économétrie spatiale soit susceptible d'améliorer l'analyse. Elles permettent en effet de traiter les grandes particularités des données spatiales : l'autocorrélation spatiale qui se réfère à la dépendance entre observations géographiques, et l'hétérogénéité spatiale qui est liée à la différenciation dans l'espace des variables et des comportements (Le Gallo, 2002). L'autocorrélation spatiale, pour une variable, permet de déterminer s'il existe une relation fonctionnelle entre ce qui se passe en un point de l'espace et ce qui se passe ailleurs. Bockstael (1996) ou encore Anselin (2001) montrent la portée que peuvent avoir de telles méthodes en économie de l'environnement, et notamment pour l'évaluation de services.

## Références

Anselin, L., 2001. Spatial effects in econometric practice in environmental and resource economics. American Journal of Agricultural Economics 83, 705-710.

Bateman, I., Brouwer, R., Ferrini, S., Schaafsma, M., Barton, D., Dubgaard, A., Hasler, B., Hime, S., Liekens, I., Navrud, S., De Nocker, L., Sceponaviciute, R., Semeniene, D., 2011. Making Benefit Transfers Work : Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe. Environmental and Resource Economics 50, 365-387.

Bockstael, N., E.,, 1996. Modeling economics and ecology : the importance of a spatial perspective. American Journal of Agricultural Economics 78, 1168-1180.

Commisariat Général du Plan, Rapport du groupe d'experts présidé par M Boiteux, 2001. Transport : choix des investissements et coûts des nuisances, Paris.

Le Gallo, J., 2002 Économétrie spatiale : l'autocorrélation spatiale dans les modèles de régression linéaire. Economie & prévision 155, 139-157.

Liekens, I., Schaafsma, M., De Nocker, L., Broekx, S., Staes, J., Aertsens, J., Brouwer, R., 2013. Developing a value function for nature development and land use policy in Flanders, Belgium. Land Use Policy 30, 549-559.

Martin-Ortega, J., Brouwer, R., Ojea, E., Berbel, J., 2012. Benefit transfer and spatial heterogeneity of preferences for water quality improvements. Journal of Environmental Management 106, 22-29.

Schaafsma, M., Brouwer, R., Rose, J., 2012. Directional heterogeneity in WTP models for environmental valuation. Ecological Economics 79, 21-31.

# Annexe A

# Complementary data

## A.1 Complementary data on soil organic carbon and vegetation carbon stocks

**Table A.1** – Mean values of soil organic carbon content in France per LULC types (from Martin et al, 2011)

| | Min. | 1st Qu. | Median | Mean | 3rd Qu. | Max. |
|---|---|---|---|---|---|---|
| Croplands | 9.92 | 39.00 | 48.00 | 51.60 | 59.90 | 137.00 |
| Forest | 15.9 | 56.4 | 75.5 | 83.8 | 106.0 | 259.0 |
| Grasslands | 11.7 | 59.6 | 79.2 | 85.2 | 101.0 | 309.0 |
| Orchards | 16.1 | 36.2 | 45.1 | 47.1 | 54.7 | 97.6 |
| Vineyards | 5.09 | 23.90 | 32.20 | 34.30 | 46.20 | 63.20 |
| Wetlands | 170 | 197 | 225 | 235 | 267 | 310 |
| Other natural areas | 2.84 | 22.20 | 23.50 | 21.80 | 26.50 | 28.90 |

# Annexe B

# Supplementary materials for the ecosystem services modeling

## B.1 Air quality regulation

Air pollutants could be removed from the atmosphere by trees in urban and peri-urban green spaces. The pollutant removal is made through dry deposition, a mechanism by which gaseous and particle pollutants are transported to, and absorbed into plant surfaces (Jim and Chen, 2008). Following Lovett et al (1994), the removed pollutant fluxes $F_i$ (in g/$cm^2$/s) of pollutant $i$ is equal to the product between the dry disposition velocity ($V_{di}$ in cm/s) and the hourly concentration of pollutant in air ($C_i$ in g/$cm^3$)as follows :

$$F_i = V_{d_i} \times C_i$$

The total pollutant flux is given by $F_i$, and is then expressed per unit area (ha) and time unit (year).

The deposition process is controlled stomata functioning depending on their density and aperture (Lovett et al, 1994). The $V_{di}$ thus represents the average rate when stomata are open, and the stomata conductance depending on various factors such as the air humidity, wind velocity, the temperature, soil moisture or the tree's health (Jim and Chen, on 2008). The calculation of air pollutant removal is based on a year time period for 12 h per day (average daytime). The $V_{di}$ are based on mean values from literature (Lovett et al, 1994 ; Jim and Chen, 2008) and are summarized in Table B.1.

**TABLE B.1** – Data for calculation of pollutant removal by trees for both study areas

| Vienne (Zone 1) | Concentration ($C_i$ g/$m^3$) | $V_{di}$ (m/s) | Pollutant removal $F_i$ (kg/ha/year) |
|---|---|---|---|
| $NO_2$ | 0,000027 | 0,0037 | 15,76 |
| $PM10$ | 0,000019 | 0,0064 | 19,18 |
| $O_3$ | 0,000063 | 0,0045 | 44,73 |
| $SO_2$ | 0,000001 | 0,0055 | 0,87 |
| $CO$ | n/a | 0 | 0 |
| Haute-Vienne (Zone 2) | Concentration ($C_i$ g/$m^3$) | $V_{di}$ (m/s) | Pollutant removal $F_i$ (kg/ha/year) |
| $NO_2$ | 0,000027 | 0,0037 | 15,76 |
| $PM10$ | 0,000019 | 0,0064 | 19,18 |
| $O_3$ | 0,000063 | 0,0045 | 44,73 |
| $SO_2$ | 0,000001 | 0,0055 | 0,87 |
| $CO$ | n/a | 0 | 0 |

We can compare these results with other studies (see table B.2), specifically with the ones of Jim and Chen (2008) estimating the pollutant removal service performed by forests in Ganzhou (China) ; Nowak (2006) for 55 cities in united states ; and Baumgardner et al (2012) for the Mexico city.

**TABLE B.2** – Pollutant removal by trees (in Kg/ha/year for three other studies

| $F_i$ (kg/ha/year) | Nowak et al, (2006) exemple for Buffalo | Jim et al, (2008) | Baumgardner et al, (2012) |
|---|---|---|---|
| $NO_2$ | 10 | 26,4 | 0,7 |
| $PM10$ | 21 | 6,1 | 71,3 |
| $O_3$ | 37 | n/a | 19,7 |
| $SO_2$ | 15 | 27,2 | 2,3 |

## B.2 Local climate regulation

The mean values we apply to cultural productions are calculated for both Zone 1 and Zone 2. Cropland productions are generally valued at their market prices, however it can be questionable to consider that the (natural) environment plays a major role in the production. Agricultural production is also supported by several human inputs : machines, human work, fertilizers, irrigation, etc. It is thus necessary to determine the contribution of the (natural) environment to the agricultural production and to value the service in consequence in order to avoid the overestimation of the service supply.

To quantify the service in physical terms, we used representative data on the agricultural production for each type of culture present in the area (in both zones) from departmental agricultural statistics (Agreste). We then used markets prices of these productions, and approximated the share of the final production that is related to human inputs (intermediate consumption, subventions), to withdraw this percentage to the final production. This percentage is calculated because there is a lack of sufficiently detailed data. We computed the percentages on the basis of the departmental agricultural accounts (here Agreste-Comptes de la nation par département 2010).

Finally, the per hectare value is computed by multiplying the yield per hectare of the cultivated area by its market price deducted from the price of intermediate consumptions and subventions. This was done for the main crop productions on the two regions, and then averaged across different crop productions. For instance, according to these accounts for Zone 1, the share of intermediate consumption and of subvention is about 82% of the total production. The average value of agricultural production of annual crops is about 1,222 euros/ha/year. The mean value applied to the ecological function sustaining annual crops was then 220 euros/ha/year for zone 1. The same computation is applied to the Zone 2 and we obtained an average value about 260 euros/ha/year (the share of intermediate consumption and of subvention is about 75% of the total production and the annual value 1,041 euro/ha/year).

## B.3 Flood protection

To model the flood protection service supplied by forests we followed Biao et al (2010), who studied the forests surrounding Beijing (China). They proposed to model the service supply from maximal potential rainfall interception during the largest rainfall event in the year. Interception and evaporation performed by forests reduce the risk of floods, while on a non-permeable soil the volume runs off immediately, forests reduce the peak flow rate at the small watershed scale due to interception. The maximum potential interception (in $m^3$) can be estimated by summing the rate of canopy interception (C), litter (L) and soil (S) retention.

C depends primarily on climatic conditions and on the rate of canopy interception depending on forest type and on the leaf area index[1]. C is the product between the interception rate per forest type $i$ ($\alpha_i$ in %) and the amount of the largest rainfall event in

[1]The LAI represents the amount of leaf material in an ecosystem and is geometrically defined as the total one-sided area of photosynthetic tissue per unit ground surface area.

the area (P in mm). L depends on the forest type, on the litter thickness layer (PL in cm) and its capacity of retention ($\beta_i$ in ton/cm/ha). Finally, S depends on the soil capillary porosity ($\mu_i$ in %) of different types of forest and on soil depth (PS in cm). Data used in calculation are presented in Table B.3.

**TABLE B.3** – Data used for the flood regulation service performed by forests

| Forest Type | Coniferous | Broadleave | Mixed forests | Shrubs | Source |
|---|---|---|---|---|---|
| $\alpha_i$ (%) | 42.21 | 46.79 | 47,39 | 42,20 | Biao et al (2010) |
| $\beta_i$ (ton/cm/ha) | 10,55 | 13,03 | 14,3 | 6,06 | Biao et al (2010) |
| PL (cm) | 7 | 2 | 2 | 2 | |
| $\mu_i$ (%) | 42.21 | 46.79 | 47,39 | 42,20 | Biao et al (2010) |
| Other data | | | | | |
| PS (cm) | | GIS data | | | Inra Infosol |
| P (Vienne) (mm) | | 92,2 | | | Meteo France |
| P (Haute-Vienne) (mm) | | 77,2 | | | Meteo France |

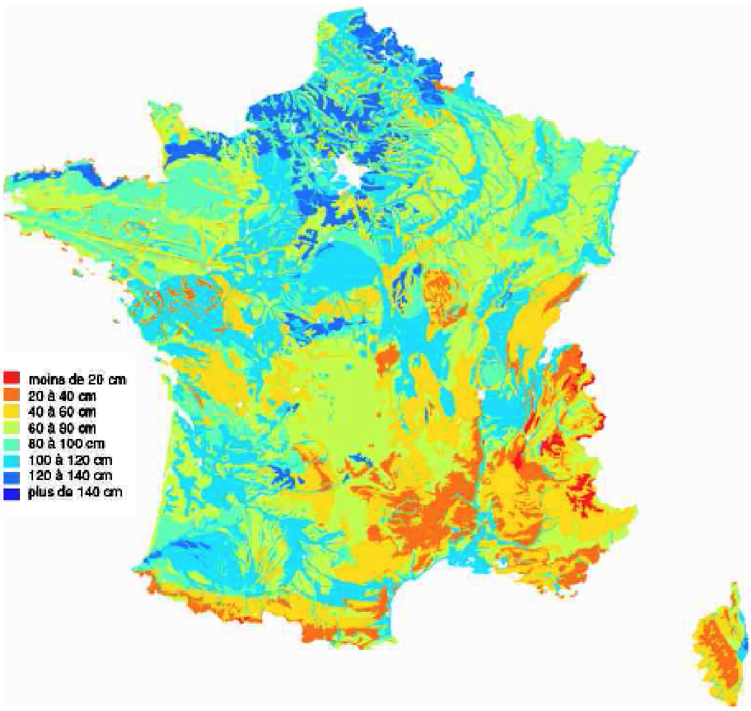



**FIGURE B.1** – Mean soil depth in France (From Inra Infosol-Gissol)

The spatial distribution of soil depth in France is represented in Figure B.1. With this calculation, we found a rainfall interception of approximately 80.8 $m^3$/ha and 68.6 $m^3$/ha respectively for the eastern and western parts (Vienne and Haute-Vienne) of the study area. Both areas contain only broadleaved forest.

## B.4 Erosion prevention

The erosion prevention service estimates the ability of a landscape to retain soil. The service is typically calculated as a function of vegetation cover, rainfall erosivity (R), slope length-gradient factor (LS) and soil erodibility (K), integrated into the Universal Soil Loss Equation (USLE)(Wischmeier and Smith 1978), which is the most often used (Crossman et al, 2013). Many studies of modeling and mapping erosion prevention exist, for example Kareiva et al (2011), Egoh et al (2008), Gascoigne et al (2011), and Nelson et al (2009). Our calculation was based on the InVEST model to assess the service supply (Kareiva et al, 2011). The model is ran with and without the infrastructure to calculate the loss of avoided erosion associated with the construction. The USLE can be written as follows (Wischmeier and Smith, 1978) :

$$USLE = R \times K \times LS \times C \times P$$

Where R is the rainfall erosivity, K is the soil erodibility factor, LS is the slope length-gradient factor, C is the crop/vegetation and management factor and P is the support practice factor.

The Slope Length Factor (LS) is one of the most critical parameters in the USLE. Slope length is the distance from the origin of overland flow along its flow path to the location of either concentrated flow or deposition. It reflects the indirect relationship between slope and land management (terracing, ditches, buffers, barriers). The steeper and longer the slope is, relative to the conditions of the reference site, the higher the risk for erosion is. In the InVEST model, different LS equations are automatically used for slope conditions (low or high). Then a Digital elevation model (DEM) is required (A GIS raster data-set with an elevation value for each cell). The Rainfall erosivity index (R) can be calculated by using the Renard and Freimund (1994) formula approximating the R factor with monthly precipitations (further information can also be found in Meusburger et al, 2012). K can be found from the Panagos et al, (2012) publication and the European Soil Portal[2]. C factor can be found in Kouli et al, (2009) and depend on land use and cover. The P factor is difficult to find, and as proposed in the user's guide, if no data are available a factor equal to 1 can be used for natural areas can be used whereas a factor equal to 0.5 can be used for managed cultures (as in Table B.4).

[2]http ://eusoils.jrc.ec.europa.eu/library/themes/erosion/Erodibility/

**Table B.4** – Data used for the flood regulation service performed by forests

| Land use | C factor | P factor |
|---|---|---|
| Broad-leaved forest | 0.13 | 1 |
| Fruit trees and berry plantations | 0.17 | 1 |
| Mixed forest | 0.18 | 1 |
| Vineyards | 0.29 | 0.5 |
| Olive groves | 0.30 | 0.5 |
| Annual croplands | 0.30 | 0.5 |
| Coniferous forest | 0.33 | 1 |
| Transitional woodland-shrub | 0.37 | 1 |
| Sclerophyllous vegetation | 0.41 | 1 |
| Complex cultivation patterns | 0.42 | 0.5 |
| Non-irrigated arable land | 0.49 | 0.5 |
| Moors and heathland | 0.50 | 1 |
| Pastures | 0.54 | 0.5 |
| Natural grasslands | 0.54 | 1 |
| Construction sites | 0.54 | 1 |
| Beaches, dunes, sands | 0.57 | 1 |
| Sparsely vegetated areas | 0.64 | 1 |
| Urban area | 0.70 | 1 |
| Bare rocks | 0.78 | 1 |

The ability of vegetation to keep soil in place on a given pixel is given by comparing erosion rates on that pixel to what erosion rates would be on that pixel without vegetation present (bare soil). The bare soil estimate is calculated as the RKLS function, that is the product between R, K and LS. Erosion from the pixel with existing vegetation is calculated by the USLE equation, and the avoided erosion on the pixel is then calculated by subtracting USLE from RKLS.

Vegetation does not only keep sediment on the pixel studied, it also intercepts sediment that has eroded upstream. The USLE equation overlooks this component of sediment dynamics, and the model takes into account this phenomena. The eroded soil estimated by the USLE equation is routed downstream via a flow-path. The model calculates how much of the sediment eroded on all pixels will be trapped by downstream vegetation based on the ability of vegetation in each pixel to capture and retain sediments.

## B.5 Pollination

The model is based on the InVEST model using a land use and land cover (LULC) map, showing both natural and managed land types. For each type of cover, the model requires estimates of both nesting site availability and flower availability (*e.g.*, for bee food : nectar and pollen). These data are expressed in the form of relative indices (between 0 and 1). A table of multiple pollinators' species has been produced for the study (123 species). The availability of nesting substrates is estimated separately for multiple nesting guilds (*e.g.*, ground nesters, cavity nesters). Foraging distances that each pollinators species typically roam is also required affecting both their persistence and the level of service they deliver to agricultural covers.

Using these data, the model first estimates the presence and abundance index of each pollinator species in every cell in the landscape, based on the available nesting sites in that cell and the flowers (food) in surrounding cells. Flowers in nearby cells are given more weight than distant cells, according to the specie's average foraging range. Since pollinator abundance is limited by both nesting and floral resources, the pollinator abundance index on cell $x$, $P_x$, is simply the product of foraging and nesting such that :

$$P_x = N_j \cdot \frac{\sum_{m=1}^{M} F_j e^{\frac{-D_{mx}}{\alpha}}}{\sum_{m=1}^{M} e^{\frac{-D_{mx}}{\alpha}}}$$

Where $N_j$ is the suitability of nesting of LULC type $j$, $F_j$ is the relative amount floral resources produced by LULC type $j$, $D_{mx}$ is the Euclidean distance between cells $m$ and $x$ and $\alpha$ is the expected foraging distance for the pollinator (Greenleaf et al, 2007 ; Londsorf et al, 2009).

The result is a map of the abundance index (0-1) for each species, which represents a map of "pollinator supply" (*i.e.*, pollinators available to pollinate crops). In this sense, the produced map represents the potential sources of pollination services, but it does not take into account yet for demand. In other words, the landscape may be rich in pollinator abundance, but if there are no bee-pollinated crops on that landscape, those bees will not be providing the service of crop pollination. To make this connection between areas of "supply" and "demand" the model calculates an abundance index of visiting bees at each agricultural cell, by again using flight ranges of pollinator species to simulate their foraging in nearby cells. Specifically, it sums pollinator supply values in cells surrounding each agricultural cell, again giving more weight to nearby cells. This sum, created separately for each pollinator species at each agricultural site, is an index of the abundance of bees visiting each farm site (*i.e.*, agricultural covers abundance). We use the foraging framework described in the previous equation to determine the relative abundance of bees that travel from a single source cell x to forage on a crop in agricultural cell $o$ :

$$P_{ox} = \frac{\sum_{m=1}^{M} P_x e^{\frac{-D_{ox}}{\alpha}}}{\sum_{m=1}^{M} e^{\frac{-D_{ox}}{\alpha}}}$$

Where $P_x$ is the supply of pollinators on cell $x$, $D_{ox}$ is distance between source cell x and agricultural cell $o$, and $D_{ox}$ is species´ average foraging distance. The numerator of this equation represents the distance-weighted proportion of the pollinators supplied by cell m that forage within cell o and the numerator is a scalar that normalizes this contribution by the total area within foraging distance. The total pollinator abundance on agricultural cell $o$, $P_o$, is simply the sum over all M cells. The second map represents the relative degree of pollination service at the demand points, or points at which this service is "delivered" : agricultural cells. A representation of the service indicator (ranging from 0 to 1) is given in figure B.2.

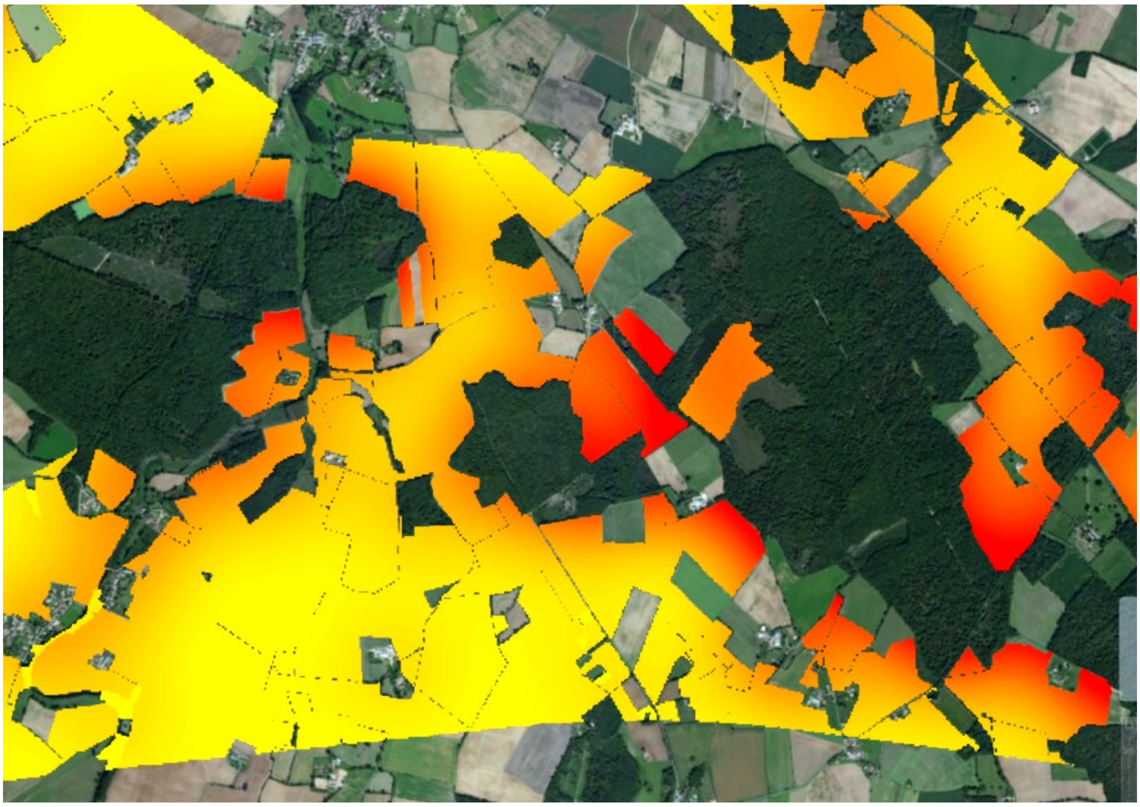


**FIGURE B.2** – Pollination service indicator (ranging from 0 to 1)

The annual economic benefit received from pollination depends on the crops dependence in each cell to pollination. Considering that the contribution of pollinators to crop yield is about 10% (Gallai et al, 2009)[3], to monetize the service we multiply the pollination score (from 0-1), to 10% of the average crop production value in the area (AGRESTE agricultural statistics (2010), excluding subventions and intermediate inputs values)[4]. We ran the model with and without the infrastructure to assess the loss associated with its construction.

## B.6 Hunting recreation

The hunting recreation service is approached by using the Optiflux Software to determine the impact of the infrastructure construction on (hunted) species habitat and migration. Optiflux is based on an evaluation of the spatial distribution of an animal population according to its ecological requirements. It requires knowledge of landscape ecology principles, such as habitats (the quality of the environment in relation to the species ecological requirements) and ecosystem functioning (the natural habitat role in the species ecology, feeding, breeding, migration, etc.). It is also based on the resistance of the natural environment to the presence of animal species : frequentation or avoidance of a natural habitat, the death rate and the energy spent in migrating within this natural habitat.

The resistance coefficients are assigned given to every type of habitat. It results in the MCR (Minimal Cumulated Resistance). The MCR gives weighted distances that are not expressed in straight lines between two points ($D_{max}$) but that reflect the resistance of the habitat crossed. The dispersion equation would have the following form :

$$MCR = D_{ij} \times r,$$

where $D_{ij}$ represent the covered distance between point $i$ and point $j$ in different habitats, and r is the resistance coefficient of every crossed habitat. If $r = 1$, then $D_{ij} = D_{max}$, if $r = 100$, then $D_{ij} = D_{max}/100$.

Dispersion rates are meaningful only if resistance has biological reality. The OptiFlux process will therefore give the user a quantitative assessment of the resistance of an existing habitat, and conversely identifies those areas where dispersion across the potential linear construction will be the greatest. Resources can therefore be targeted to maintain

[3] If more detailed data on agricultural cover are available, coefficient from Klein et al (2007) should be used.

[4] It is the same agricultural output then the on calculated for the local climate regulation. We estimated also the value for orchard and vineyards by applying the same methodology.

or improve the connectivity of these locations and optimize biological function of existing ecosystems. A representation of the loss considered for Zone 1 calulated with optiflux is given in Figure B.3.

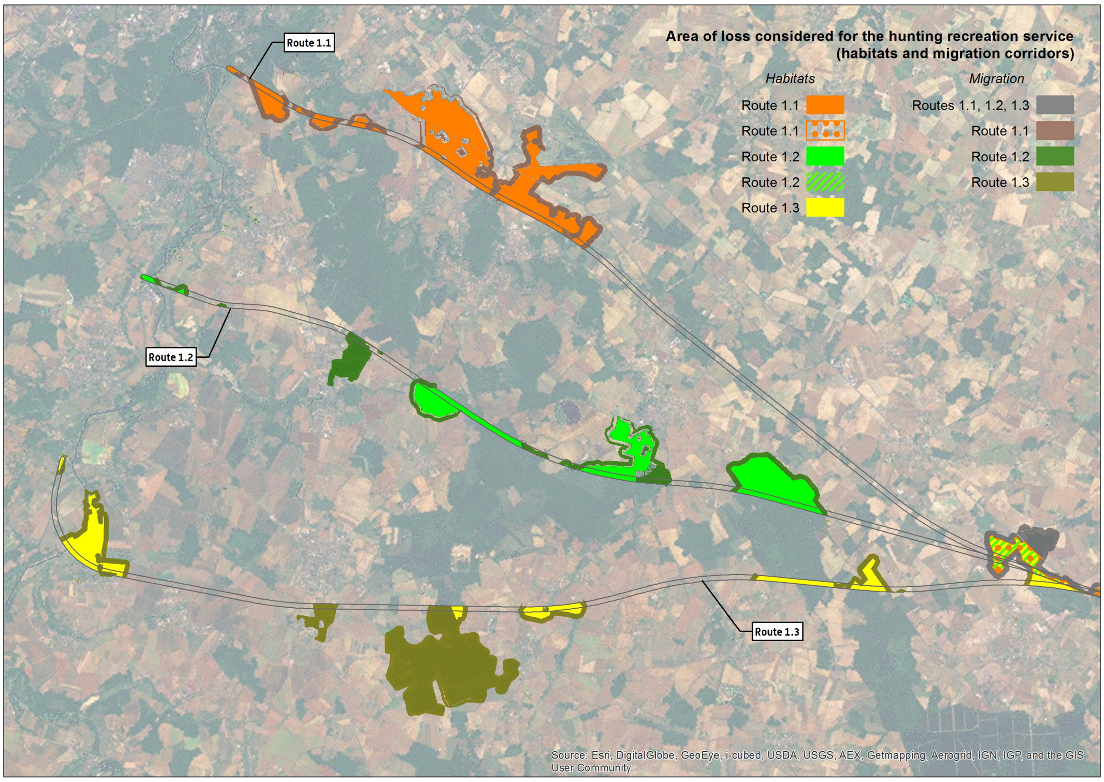



**FIGURE B.3** – Area of loss considered for the hunting recreation service (habitats and migration corridors) for Zone 1

# References


AGRESTE, 2010. Memento de la production agricole. (Online) Available from : `http://agreste.agriculture.gouv.fr/`.

Baumgardner, D., Varela, S., Escobedo, F.J., Chacalo, A., Ochoa, C., 2012. The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. Environmental Pollution 163, 174-183.

Biao, Z., Wenhua, L., Gaodi, X., Yu, X., 2010. Water conservation of forest ecosystem in Beijing and its value. Ecological Economics 69, 1416-1426.

Crossman, N.D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., Drakou, E.G., Martìn-Lopez, B., McPhearson, T., Boyanova, K., Alkemade, R., Egoh, B., Dunbar, M.B., Maes, J., 2013. A blueprint for mapping and modelling ecosystem services. Ecosystem Services 4, 4-14.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. Agriculture, Ecosystems & Environment 127, 135-140.

Gallai, N., Salles, J.-M., Settele, J., Vaissière, B.E., 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. Ecological Economics 68, 810-821.

Gascoigne, W.R., Hoag, D., Koontz, L., Tangen, B.A., Shaffer, T.L., Gleason, R.A., 2011. Valuing ecosystem and economic services across land-use scenarios in the Prairie Pothole Region of the Dakotas, USA. Ecological Economics 70, 1715-1725.

Jim, C.Y., Chen, W.Y., 2008. Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China). Journal of Environmental Management 88, 665-676.

Kareiva, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S.E., 2011. Natural capital : Theory and practice of mapping ecosystem services. Oxford University Press, Oxford.

Kouli, M., Soupios, P., Villianatos, F., 2009. Soil erosion prediction using the Revised Universal Soil Loss Equation (RUSLE) in a GIS framework, Chania, Northwestern Crete, Greece. Environmental Geology 57, 483-497.

Lovett, G.M., 1994. Atmospheric deposition of nutrients and pollutants in north America : An ecological perspective. Ecological Applications 4, 629-650.

Meusburger, K., Steel, A., Panagos, P., Montanarella, L., Alewell, C., 2012. Spatial and temporal variability of rainfall erosivity factor for Switzerland. Hydrol. Earth Syst. Sci. 16, 167-177.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7, 4-11.

Nowak, D.J., Crane, D.E., Stevens, J.C., 2006. Air pollution removal by urban trees and shrubs in the United States. Urban Forestry and Urban Greening 4, 115-123.

Panagos, P., Meusburger, K., Alewell, C., Montanarella, L., 2012. Soil erodibility estimation using LUCAS point survey data of Europe. Environmental Modelling & Software 30, 143-145.

Renard, K. G., Freimund, J. R., 1994. Using monthly precipitation data to estimate the R-factor in the revised usle. Journal of hydrology 157, 287-306.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses-a guide to conservation planning. US Department of Agriculture Science and Education Administration, , Washington, DC.

# Annexe C

# Non-market valuation methods to assess time savings and environmental external effects

**Times savings :**

time savings constitutes a utility gain for travelers varying in function of the travel purpose (labor, leisure), users' wages, transport mode, distance, duration, and comfort. Reference values (in €/h/person and per trip) are based on literature reviews and are assumed to reflect observed behaviour and then users' willingness to pay. In urban areas, the mean value varies according to trip purposes. For intercity journeys, mean values vary according to distance and transport mode. The number of users benefiting from these gains, and the time saved by users are computed by relying on traffic models. Time saving values are presented in Table C.1.

**Greenhouse gas emission reduction :**

(only for $CO_2$ emission) is valued in the same way as in the Chapter 2 (based on the Quinet report (2009) which relies on a cost-effectiveness analysis). The emissions value computed for the operating infrastructure are expressed in €/vehicle/km and are differentiated by transport mode, and distance traveled ($<$50 km, between 50 and 150 km, $>$ 150 km).

**Road insecurity :**

is based on the "value of life" computed with stated preferences methods. The objective is to protect the maximum of human lives accounting for budget constraints, and to distribute

**TABLE C.1** – Time savings values (in euros 2000/hour)-source Instruction Cadre (2005)

| | Travelers in urban areas (in euro 2000/h) | |
|---|---|---|
| Travel purpose | France | Ile de France |
| Professional travel | 10.5 | 13 |
| Residence-work travel | 9.5 | 11.6 |
| Various travel (shopping, leisure, tourism, etc) | 5.2 | 6.4 |
| Average value | 7.2 | 8.8 |

| | Travelers in inter-urban areas (in euro 2000/h) | |
|---|---|---|
| Travel type | Value varying with the distance | Stabilisation for distances over 400 km |
| Road | 8.4 to 13.7 | 13.7 |
| Train (2nd class) | 10.7 to 12.3 | 12.3 |
| Train (1st class) | 27.4 to 32.3 | 32.3 |
| Air | 45.7 | 45.7 |

security efforts until full equalization of avoidance costs of an additional death. According to this approach, a public program is considered optimal regarding security when the cost committed to save an additional life is equal to the community average marginal willingness to pay for security : this leads one to determine this value by surveys, either with the decision-makers, or with a population sample revealing the community position.

**TABLE C.2** – Value retained for a human life saved (in Million Euros 2000)-source Instruction Cadre (2005)

| Basic value | Collective transport | Road transport |
|---|---|---|
| Killed (in million Euros 2000) | 1.5 | 1 |
| Seriously injured (in Euros 2000) | 225,000 | 150,000 |
| Slightly injured (in Euros 2000) | 33,000 | 22,000 |

### Atmospheric pollutants :

(PM10, $SO_2$, CO, NOX, volatile organic compounds (VOC)) emissions are valued based on the effect on mortality and morbidity. Emissions depend on the transport mode, and n the type of vehicle. Values are based on a percentage of the "value of life" (reduction in the average statistical life expectancy, or reduction on death risk) applied to death caused by cardiovascular or respiratory distress and by lung cancer. Values are differentiated according to population density in the polluted zones (3 classes : dense urban area (>420 inhabitants per $km^2$), diffuse urban area and rural areas (<37 inhabitants per $km^2$). Values for atmospheric pollution impacts are presented in Table C.3.

**Table C.3** – Atmospheric pollution (in euros 2000/100.vehicle-km)-source Instruction Cadre (2005)

| Road (in euros 2000/100.vehicle-km) | Dense Urban | Diffuse Urban | Open country | Weighted average |
|---|---|---|---|---|
| Private cars and light commercial vehicles | 2.9 | 1 | 0.1 | 0.9 |
| Heavy trucks | 28.2 | 9.9 | 0.6 | 6.2 |
| Bus | 24.9 | 8.7 | 0.6 | 5.4 |
| Rail (in euros 2000/100.vehicle-km) | Dense Urban | Diffuse Urban | Open country | Weighted average |
| Diesel train (freight) | 458 | 160 | 11 | 100 |
| Diesel train (passengers) | 164 | 57 | 4 | 36 |

### Traffic congestion :

is valued as the difference between the utility derived from the actual use of the congested infrastructure and the utility the consumer would derive if it were used in an optimal way. Time gains are derived from traffic decongestion. The value is differentiated according to population density.

### Noise disturbance :

expresses the damage borne by the local residents. The effect is valued via a series of values based on the depreciation of the average rent per square meter of surface occupied and exposed to noise levels exceeding a certain threshold (value transfer based on a hedonic pricing study expressed in % of rent depreciation/ DB(A)). Further, values are differentiated to take the nature of concerned zones into account(housing, leisure, public institutions). An increase is applied to the value when the noise exceeds 70 DB(A) to account for the long term effect on health. Values vary according to the transport mode and distances travelled. Values retained by the Boiteux II report (2001) are presented in Table C.4.

**Table C.4** – Noise disturbance values (in % of depreciation of average rents per square metre of surface occupied)-source Instruction Cadre (2005)

| Exposure to sound (in dB) | 55-60 | 60-65 | 65-70 | 70-75 | + than 75 |
|---|---|---|---|---|---|
| % depreciation/decibel | 0.4% | 0.8% | 0.9% | 1% | 1.1% |

These external effects valued in €/per traveler per kilometer or in €/vehicle per kilometer, are based on traffic models. Traffic models predict the traffic according to the modal report predicted and induced traffic (passengers who did not travel before, or who travelled less).

## Abstract: Integrating ecosystem services in the evaluation of transport infrastructure projects

The purpose of this thesis is to broaden the assessment process of terrestrial transport infrastructure in the field of Ecosystem Services (ES), i.e., the benefits people derive from ecosystems. To achieve this, we first review the major challenges to integrate the ES approach into transport infrastructure decisions. This inclusion is only possible if changes in ES are explained in a spatially explicit way (Chapter 1). We illustrate this point by assessing the loss of the global climate regulation service caused by the infrastructure construction (Chapter 2). The analysis is based on the examination of a contemporary infrastructure project in Western France, and the same case study is used in the next part of this thesis. We further deepen the issue of combining direct loss of multiple ES with indirect loss due to the infrastructure impacts on landscape connectivity (Chapter 3). For both direct and indirect effects we integrate potential threshold effects on ES loss. We compare implementation options to provide an example of how choices can be improved by mapping ES loss associated with a combination of direct and indirect impacts. Finally, we provide a test of the usefulness of the ES consideration into environmental impact assessment and cost-benefit analysis in order to assess the additional information it may bring (Chapter 4). We show that this analysis can provide guidance at different stages of transport project: from the preliminary studies to the study of the final implementation option. For environmental impact assessment, the consideration of ES opens the possibility of measuring ES loss providing a means for selecting among a set of route options for the infrastructure. For cost-benefit analysis, since ES loss induced by the selected route is expressed in monetary terms, it can be integrated as a standard social cost in the analysis, allowing a more efficient control of natural capital loss. As a result, this may help project stakeholders to better consider the effects of the infrastructure implementation.



**Keywords:** Ecosystem services; Terrestrial transport infrastructures; Environmental impact assessment; Cost-benefit analysis; Economic valuation; Spatial assessment.



## Résumé : L’intégration des services écosystémiques dans l’évaluation des projets d’infrastructures de transport

L'objectif de cette thèse est d’intégrer la notion de Services Écosystémiques (SE), i.e., les bénéfices que la société retire du fonctionnement des écosystèmes, dans le cadre des procédures d'évaluation des projets d'infrastructures de transports terrestres. Pour cela, nous commençons par mettre en lumière les différents défis associés à l'intégration des SE dans les décisions d'implantation d'infrastructures de transport. L'intégration ne peut être réalisée que si l'estimation de la perte de SE est faite de manière spatialement explicite (Chapitre 1). Puis, nous illustrons ce point à travers l'étude de la perte d'un service : la régulation du climat global (Chapitre 2). L'analyse est basée sur l'examen d'un projet d'infrastructure contemporain dans l’ouest de la France, et le même cas d’étude est utilisé dans la suite de cette thèse. À la suite, nous approfondissons la question de la combinaison de la perte directe et de la perte indirecte de SE due aux impacts de l'infrastructure sur la connectivité des entités spatiales (Chapitre 3). Pour les deux types d'impacts, nous intégrons des seuils potentiels sur la fourniture de services en proposant une méthode de prise en compte des effets sur des écosystèmes particulièrement sensibles. Nous comparons différentes options de tracé afin de donner un exemple de la manière dont les choix pourraient être améliorés en cartographiant les pertes directe et indirecte de SE. Enfin, nous montrons l'intérêt de la prise en compte des SE dans l'étude d'impact environnemental et le bilan socio-économique de manière à mesurer l'information supplémentaire qu’apporte une telle intégration (Chapitre 4). Nous montrons que ce type d'analyse peut orienter différentes étapes d'un projet d'infrastructure, des études préliminaires jusqu’à l'étude du tracé final. Dans le cas des études d'impact environnemental, l'intégration de ces considérations permet de mesurer la perte de services engendrée par chaque tracé d'infrastructure et d'intégrer ces pertes en tant que nouveau critère de choix de tracé. Concernant le bilan socio-économique, la perte de services exprimée en termes monétaires permet de donner des informations quant à la perte sociale engendrée par le tracé final. Ceci peut aider les parties prenantes des projets à mieux appréhender les effets engendrés par la réalisation de l’infrastructure.



**Mots clés :** Services écosystémiques, Infrastructures de transport terrestres, Étude d'impact environnemental, Analyse coût-avantage, Évaluation économique, Évaluation spatiale.