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

Léa Tardieu[1*], Sébastien Roussel[1, 2], John D. Thompson[3], Dorothée Labarraque[4], Jean-Michel Salles[5]

**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 paper 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. 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 and among different types of losses because some ES, in addition to losses that are directly proportional to the surface impacted, can show additional indirect losses associated with landscape connectivity. In addition, we propose a method for assessing threshold effects in particular ecosystem types that may be 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.

**Highlights**:

► The loss of ecosystem services is rarely assessed in environmental impact assessment
► We combine direct loss with indirect loss due to impacts on landscape connectivity
► We integrate potential threshold effects on ecosystem services loss
► We compare implementation options to examine how choices can be improved

**Keywords:** Ecosystem services, environmental impact assessment, landscape connectivity, linear infrastructure, threshold behavior.

JEL Classification: H43, H54, Q51, Q57.

[1] UMR 5474 LAMETA, Université Montpellier 1, Avenue Raymond Dugrand, F-34960 Montpellier, France.
* *Corresponding Author* : Present address: 2 Place Pierre Viala (Bat 26), 34060 Montpellier, France.
E-mail adress lea.tardieu@supagro.inra.fr (L. Tardieu). Tel: +33 4 99 61 27 23
[2] Université Montpellier 3 Paul Valéry, Route de Mende, F-34199 Montpellier, France
[3] UMR 5175 CEFE, CNRS, 1919 Route de Mende, BP 5051, F-34033 Montpellier, France
[4] Egis Structures et Environnement, 33-43, Avenue Georges Pompidou, F-31130 Balma, France
[5] UMR 5474 LAMETA, CNRS, 2 Place Viala, Bâtiment 26, F-34060 Montpellier, France

## 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 20$^{th}$ 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). However, 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 the decision-making processes (Laurans et al, 2013).

The evaluation of ES loss could be particularly useful for decision-making during the comparison of alternative land development propositions. 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). In cases where projects affect areas of less remarkable biodiversity, that do not contain the emblematic or protected habitats and species that currently provide a basis for avoiding, reducing or compensating effects of development projects, such short-term economic criteria are most often given priority over environmental concerns. Assessing ES loss could thus allow for a broader identification of 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, Tardieu et al, 2013).

The impacts of land conversion 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 Impact Assessment (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 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 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 functions | References |
|---|---|---|
| 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) |
| 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) |
| Recreation | Importance of landscape diversity, impacts on particular elements of a landscape that alter the whole landscape | van der Zee (1990) |

In addition, species and ecosystems often need a minimum surface area to complete their ecological functions. Indeed, 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 behavior (Groffman et al. 2006; Lindenmayer and Luck 2005; Muradian, 2001; Swift and Hannon, 2010). Such threshold behavior is observed when the response of an ecological factor (individual behavior,

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, if a particular habitat patch plays 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 induce 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 other situations (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 behavior is a major challenge to our efforts to assess ES loss associated with development projects.

In this study, 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 ES loss for different route options for a high-speed rail project[1] in France. In addition to accounting for direct losses of ES provision due to the conversion of ecosystems, we integrate the possibility of indirect losses associated with landscape connectivity. 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, waterways) by adapting land takes to the local ecological and landscape context.

In Section 2, we present the methodological options we adopt in order to assess direct and indirect ES loss. In Section 3, we present the results in our case study of high-speed rail project implementation. We provide a discussion in Section 4 and a conclusion in Section 5.

## 2. Methodological options and data collection

The general framework for analysis is displayed in Figure 1 and the valuation steps adapted to the infrastructure construction are described in the following sub-sections.

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

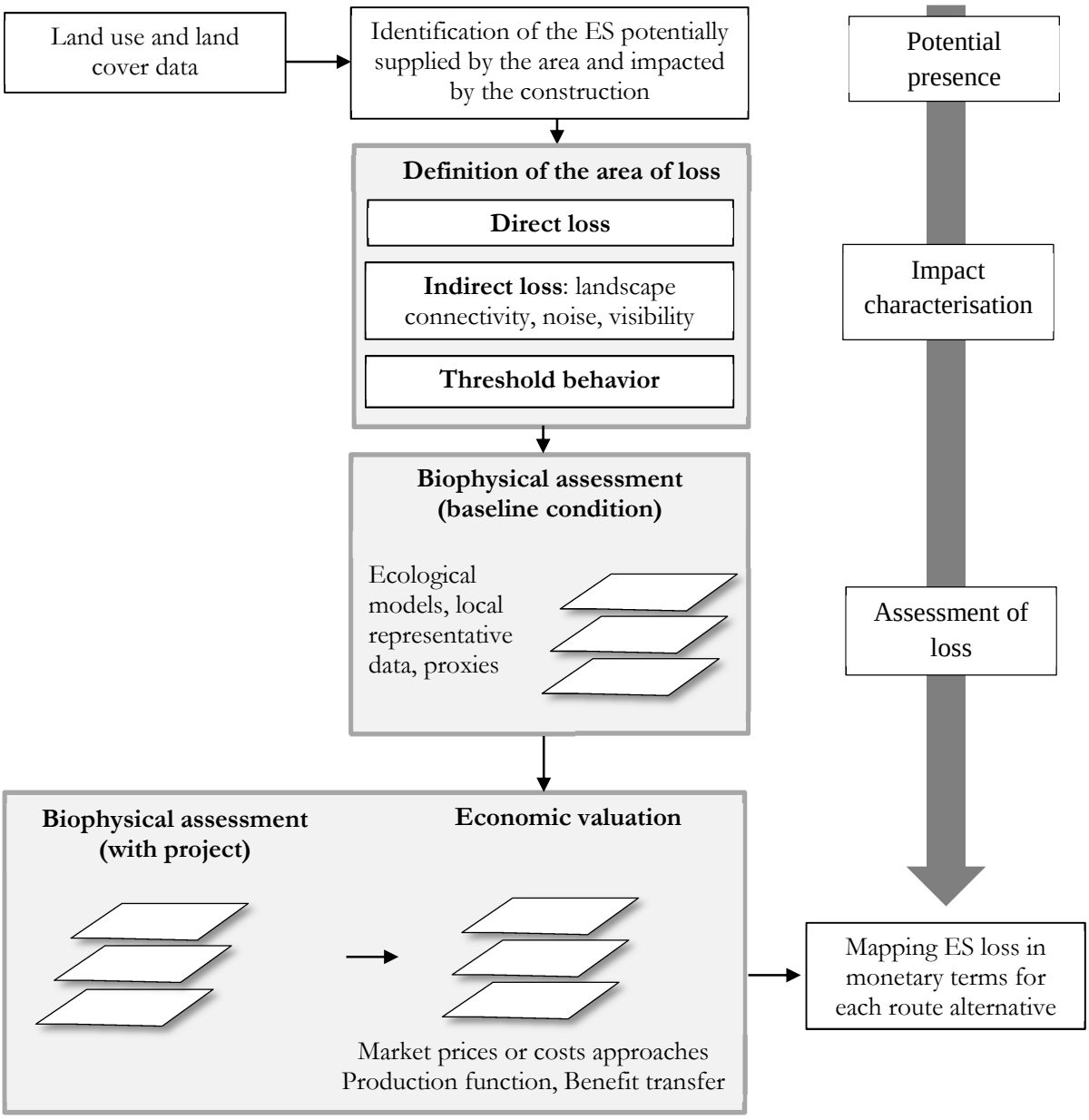


**Fig. 1.** Methodological framework summary

### 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 evaluations are required in the EIA framework.[2] We also 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 economically 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.[3] Instead, food and timber 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

[2] 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, 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.

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

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 Table 2.

**Table 2**
Classification of ES supplied and impacted for each land cover type

This classification is set 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 and soil erodability, conservation practices and vegetation retention efficiency. *Pollination*: for entomophilous crops needing insect pollination. *Fishing*: for wetlands adjacent to a river. *Recreation*: for natural areas situated in the proximity to a city.

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 (see Table 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).

### *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 to infrastructure construction which integrates direct losses and indirect losses related to landscape connectivity (Tables 1 and 2). To illustrate the effect of an indirect impact on landscape connectivity, Figure 2 represents the impact extent of the infrastructure construction on game species. Hence, in this area the service of hunting recreation can be presumed as a loss after the infrastructure construction.

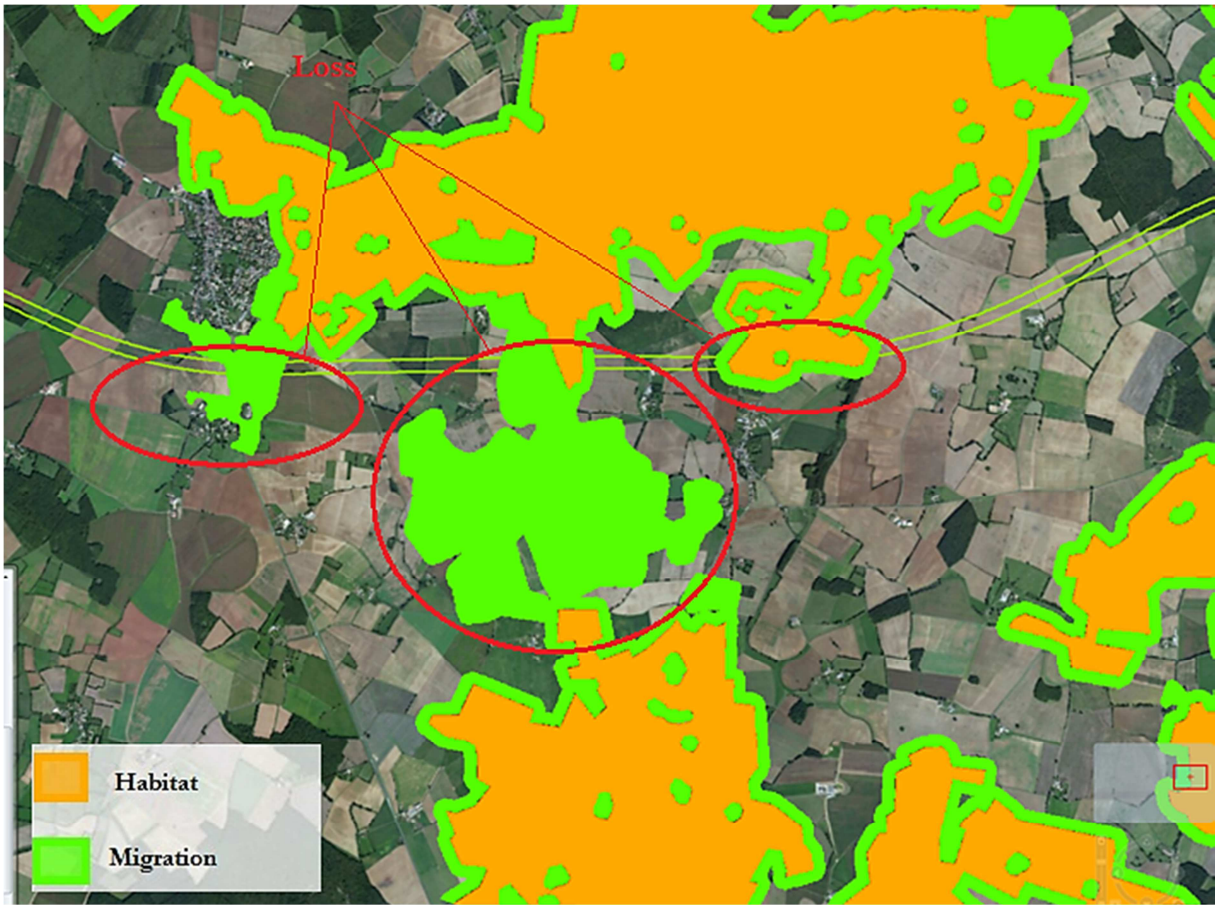


**Fig. 2.** Representation of species habitats and migration corridors threatened by the infrastructure construction

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 behavior 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 behavior 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 ecosystems 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 more generally be affected by threshold effects associated with land use transformations (King et al, 2005).

To integrate potential threshold behavior 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 a greatly extended effect of this type of impact has been demonstrated (Findlay and Bourdages, 2000; Forman and Deblinger, 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.

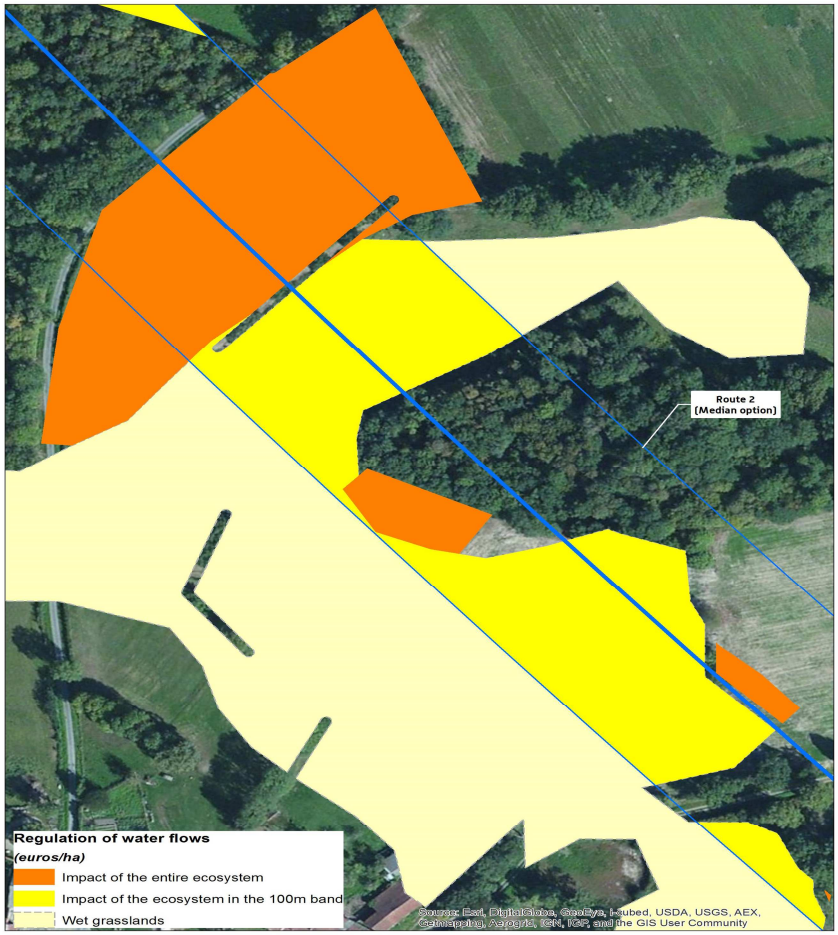


**Fig. 3.** Representation of direct and indirect ES loss for wet grasslands due to a threshold behavior assumption. The different patches of wet grasslands are considered as impacted in different ways: the patch can be totally impacted because more the 50% of its surface area is converted (dark orange) or the loss may be limited to the 100m buffer zone because less than 50% of its surface area is converted.

Outdoor recreation (picnic, ride, hike, etc) is a particular case, because the service is not necessarily 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, infrastructure visibility may cause a loss of landscape interest, and noise disturbance can cause the loss of the service over an extended area (Table 2). The impact on a recreational area will thus depend on its distance from the infrastructure.

### *2.3 Biophysical measurement of ecosystem service loss and economic values associated*

We illustrate our approach in a study area concerned by an infrastructure project in France that crosses a principally rural territory with and natural and semi-natural areas. The 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 costs, damage costs. Finally, we used present values standardized in euros for the year 2010 (i.e., we deflated 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 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 mushrooms for self-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 in France (19 Mm3/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 €/$m^3$.

For fodder provisioning 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 human health (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 µm (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).

### 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. We assessed this service by using the avoided air conditioning cost (i.e., 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 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.

### Global climate regulation

The service is assessed as the carbon storage service loss due to soil sealing, (term used by Hicks et al, 2002) 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 Tardieu et al. (2013). 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) €/m3/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).

**Water flow regulation**

Wetlands act like sponges or reservoirs, mitigating floods but also supporting river flows during the low water period. In the absence of local data on the quantification of this service, assumptions are made to estimate the volume of water involved in this phenomenon. Based on a French study in a similar area (Agence de l'Eau Adour-Garonne, 2009), we estimated volumes between 5,000 and 10,000 m3/ha for the years when low water flow replenishment occurs (approximately every 5 years). We valued this service with a replacement cost method in which the value of the service represents the

expenses necessary to replace the ecological function. The cost of destocking to support low water by the major French electric company (Electricité de France (EDF)) is about 0.0406 €/m3. The service is then evaluated at 61 €/ha/year, and is applied to wetlands directly related to a river system.

**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 erodability, 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[4] 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.

**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

[4] Integrated Valuation of Environmental Service and Trade-offs (InVEST) developed by the Natural Capital Project: http://www.naturalcapitalproject.org/InVEST.html

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 subsidies 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.

**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 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 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/year, 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 evaluations for UK and USA (Alvarez-Farizo et al. 1999 and Bergstrom 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.

### *2.4 Assessment of loss*

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

$$V(ESL) = \sum_i \sum_k D(LC_{ki}) \times v(ES_{ki}) \times p(ES_{ki}) + \sum_i \sum_k I(LC_{ki}) \times v(ES_{ki}) \times p(ES_{ki})$$

In this equation, $D(LC_{ki})$ is the area considered as directly impacted for each land cover type *i* ($i= 1,\ldots, 17$) and ES type *k* ($k= 1,\ldots, 15$), 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 economic 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. 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. 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. Results

Our study considers different route options in two zones crossed by a high-speed rail project in France. The route options 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 (routes 1.1, 1.2, and 1.3), the second zone two alternative routes (routes 2.1 and 2.2). 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 4). The longest route is route 1.3 for the first zone, and route 2.2 for the second zone.

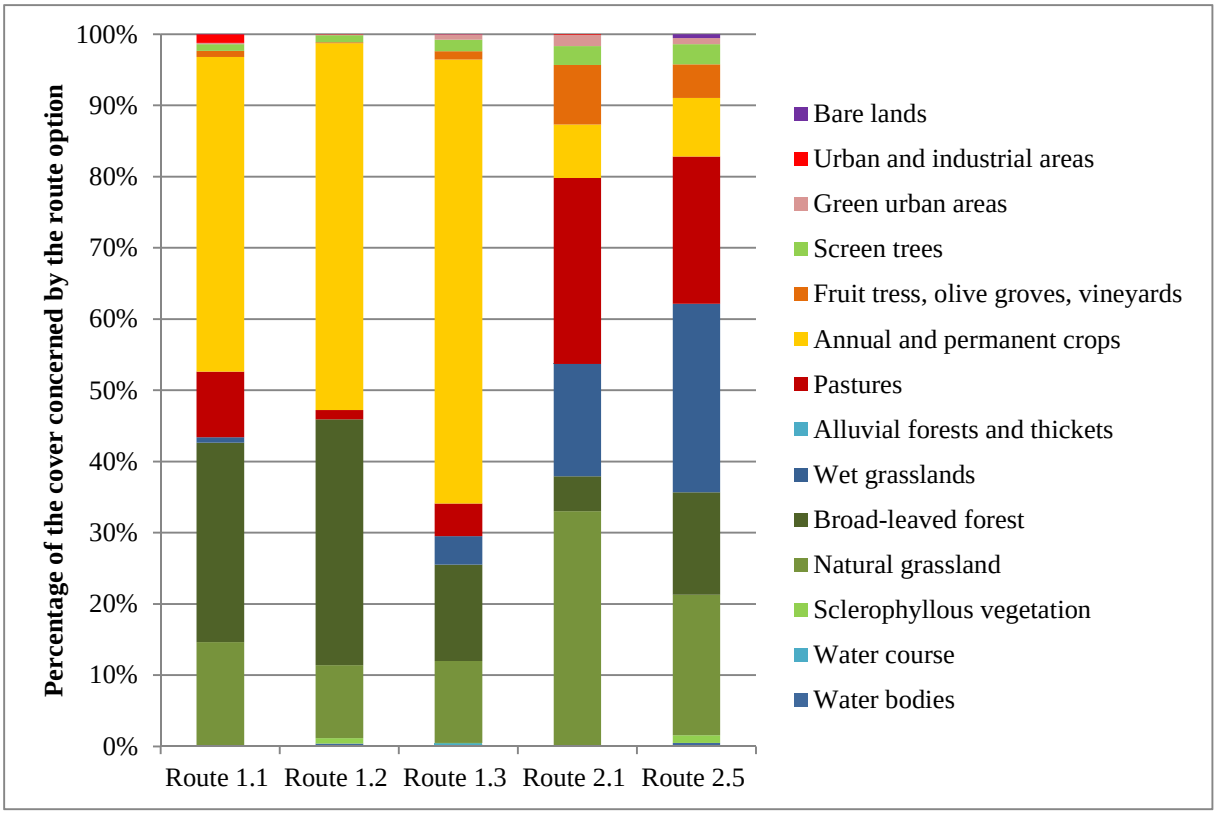


**Fig. 4**. Share of ecosystem impacted by different route options (with a 100m land take assumption)

The route options produce different ES losses (Table 4).

**Table 4**
Annual economic loss per service and per route option (in Euro 2010) (shaded cells highlight the least impacting routes)

| | | ZONE 1 | | | ZONE 2 | |
|---|---|---|---|---|---|---|
| | | Route 1.1 | Route 1.2 | Route 1.3 | Route 2.1 | Route 2.2 |
| Provision | 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* |
| Regulation | 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* |
| Cultural | 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* |
| | Length (km) | 20 | 20 | 22 | 21 | 23 |
| | **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

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 we assess the result per kilometre. 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 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 228,000 Euros (in Euro2010), while route 1.2 involves a loss of 293,000 Euros.

For zone 2, the least impacting route is route 2.1, which causes an annual loss of 245,000 euros (route 2.2 involves a loss of 282,000 euros 2010). This result is also true for the loss per kilometre. 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 2.2 (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 land use land cover 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% (Figure 5) less loss than with the more precise land cover data. 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 meaningful in estimating raw materials service, local climate regulation service, water flows, fishing recreation, or pollination service.

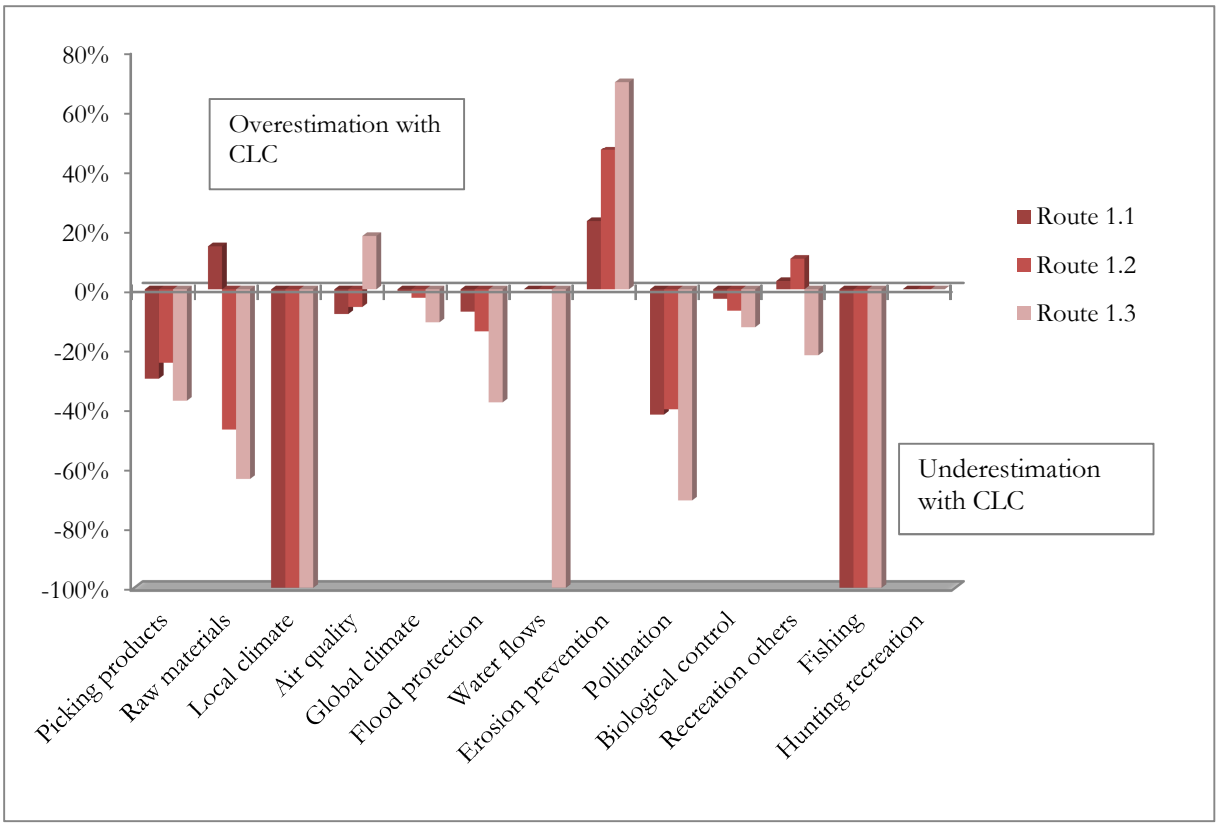


**Fig. 5.** Differences between an estimation of ES loss with the Corine Biotope Classification (scale 1:5000) and Corine Land Cover data (scale 1:25000) for the first zone

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 5). 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.

**Table 5:** Ranking sensitivity to mean values: annual loss when estimates are made with the lower combination and higher combination of biophysical and economic values (shaded cells highlight the least impacting routes)

| | **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 | 225 276 | 453 775 | 235 323 | 480 302 | 173 858 | 423 923 |
| Route lenght (km) | 20.20 | 20.20 | 20.01 | 20.01 | 22.34 | 22.34 |
| 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 | 229 807 | 471 680 | 248 145 | 525 971 |
| Route lenght (km) | 21.59 | 21.59 | 22.63 | 22.63 |
| Annual loss/km | 10 644 | 21 847 | 10 965 | 23 242 |

## 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 analyzed here are not currently integrated into environmental impact assessment. Our study thus provides an initial attempt to integrate such services. In addition we do so in a way which points out the necessity of discriminating and combining 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 behavior 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.

### 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; the overall least impacting route option (route 1.3) is the option with the highest 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 due to the important effects on landscape connectivity. This result may be even more critical if particular habitats that are impacted play a key role in movement patterns (Gaaff and Reinhard 2012), particularly for the movements of large mammals (Forman and Deblinger, 1999; Hilty et al, 2006) or species that incur high dispersal mortality (Flather and Bevers 2002).

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).

Finally, it should be noted that we based our study on gross impacts, i.e., we did not take into account mitigation measures such as wildlife passageways. This is because at this stage of the project (comparison of route alternatives), we cannot identify where or how many passageways will be implemented. We examined the introduction of a passageway for the route option 1.3 (zone 1). The passageway was placed at the location which maximizes the restoration of an ecological corridor (determined by Optiflux). This resulted in a reduction of ES loss amounting to around 17,000 euros, a considerable reduction in loss for the recreation service. Hence, once the location of passageways is known it should be integrated into the analysis of the ES loss.

### 4.2 Threshold behavior

By testing the effects of incorporating a scenario in which small scale ecosystems show a rapid threshold behavior 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 behavior 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 behavior 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).

### 4.3 Stakeholder consultation

Examined in the light of results used for the choice among different route options, our 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 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 route option involves passing near a relatively large provincial town, thus engendering noise effects and other nuisances, whilst route 2 engenders more environmental effects, a result confirmed by the assessment of ES loss. However, in zone 2, our results also showed that route

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, integrating ES loss can provide novel and complementary information for assessing environmental impact and decision making.

### 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. This approach has been criticised 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.

## 5. Conclusions

ES losses are currently poorly assessed and valued in monetary terms in EIAs for infrastructure construction. 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.

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 behavior 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 behavior, that some consideration of such threshold behavior 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 evaluation 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.

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