Environmental Impact Assessment Review
23 (2003) 343–365

Environmental Impact Assessment Review

www.elsevier.com/locate/eiar

Biodiversity Impact Assessment of roads: an approach based on ecosystem rarity

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Received 1 July 2002; received in revised form 1 November 2002; accepted 1 November 2002

Abstract

Biodiversity has become one of the central environmental issues in the framework of recent policies and international conventions for the promotion of sustainable development. The reduction of habitat worldwide is currently considered as the main threat to biodiversity conservation. Transportation infrastructures, and above all road networks, are blamed for highly contributing to the decrease in both the quantity and the quality of natural habitat. Therefore, a sound Biodiversity Impact Assessment (BIA) in road planning and development needs to be coupled to other commonly considered aspects. This paper presents an approach to contribute to BIA of road projects that focuses on one type of impact: the direct loss of ecosystems. The first step consists in mapping the different ecosystem types, and in evaluating their relevance for biodiversity conservation. This is based on the assessment of ecosystem's rarity. Rarity is a measure of how frequently an ecosystem type is found within a given area. Its relevance is confirmed by the fact that the protection of rare ecosystems is often considered as the single most important function of biodiversity conservation. Subsequently, the impact of a road project can be quantified by spatially computing the expected losses of each ecosystem type. To illustrate the applicability of the methodology, a case study is presented dealing with the assessment of alternative routes for a highway development in northern Italy.

Keywords: Biodiversity; Ecosystem; Road; Rarity; Space occupation; Impact

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0195-9255/02/$ – see front matter
doi:10.1016/S0195-9255(02)00099-9

1. Introduction

The conservation of biological diversity (biodiversity) has recently emerged as one of the major global environmental concerns (Noss and Cooperrider, 1994; George, 1999; Diamantini and Zanon, 2000). Consequently, a thorough treatment of the effects of developments on biodiversity is to be included in the procedure of Environmental Impact Assessment (EIA), as recommended by the Convention on Biological Diversity: "...each Contracting Party shall...introduce appropriate procedures requiring environmental impact assessment of its proposed projects that are likely to have significant adverse impact on biological diversity..." (UNCED, 1992, article 14). Following this recommendation, several governmental agencies have issued guidance on EIA and biodiversity (Canadian Environmental Assessment Agency, 1996; CEQ, 1993) and work is being carried out in this area also by a range of non-governmental bodies, such as the International Association for Impact Assessment (IAIA, 2001) and The World Conservation Union (Byron, 1999). This has lead to the establishment of a specific disciplinary field, namely Biodiversity Impact Assessment (BIA), which aims at developing and applying strategies for performing the analysis of the impacts on biodiversity within EIA. However, BIA can still be considered in its infancy, especially for what concerns real applications (Atkinson et al., 2000).

The decline of habitat is globally recognized as the current main threat to the conservation of biodiversity (EPA, 1999). The most severe habitat reduction occurs when a natural ecosystem is converted to an artificial system, as it happens for a road construction. Roads represent one of the most widespread forms of modification of the landscape that occurred during the past century, and particularly after World War II (Trombulak and Frissell, 2000; Smith, 1990). Road developments affect and modify the habitat conditions, which in turn influence the abundance and distribution of plant and animal species, i.e., the biodiversity, of the impacted areas. Reviews on this topic (Byron et al., 2000; Thompson et al., 1997) suggested that road schemes, due to their linear structure, are much more likely to affect natural areas than other developments in general. The dimension of the phenomenon can be pictured by recalling that many industrial nations have given about 1% to 2% of their land to roads and roadsides (Forman, 2000; Seiler and Eriksson, 1995), making the road network a common feature of virtually every landscape.

Roads cause both a direct and an indirect loss of habitat. The direct loss refers to the reduction of the total area of an ecosystem caused by the presence of the road and its verges, i.e., by the conversion of the original land cover (e.g., woodland, grassland, wetland, etc.) into an artificial surface. The indirect loss refers to effects such as the fragmentation (i.e., the portioning of an ecosystem into smaller and more isolated patches) and the degradation of ecosystems (i.e., the biophysical alteration of an ecosystem induced by noise, air and water pollution, artificial light, etc.). These effects cause an indirect loss of habitat in that they reduce the capability of an ecosystem to sustain its original biodiversity.

Comprehensive review on the ecological impacts of roads can be found in Trombulak and Frissell (2000), Forman and Alexander (1998), Southerland (1995).

This paper presents an approach to contribute to BIA of road projects that focuses on one type of impact: the direct loss of ecosystems. This is the most evident ecological effect of a road cutting through the landscape. Despite this, it is still often neglected or ill-addressed during the EIA (Geneletti, 2002; Treweek et al., 1993; Byron et al., 2000). A sound approach to assess the ecosystem-loss impact requires the preliminary setting-up of guidelines to map the different ecosystem types and to evaluate their relevance for biodiversity conservation. Subsequently, the impact of the project under study can be quantified by spatially computing the expected losses of each ecosystem type. The ecosystem's relevance is to be assessed by referring to explicit evaluation criteria. In the methodological approach described in this paper, it is proposed to resort to the criterion most frequently used in ecological evaluations: rarity (Smith and Theberge, 1986). Rarity is a measure of how frequently an ecosystem type is found within a given area. Its relevance is confirmed by the fact that the protection of rare ecosystems is often considered as the single most important function of biodiversity conservation (Margules and Usher, 1981).

To illustrate the applicability of the methodology, a case study is presented dealing with the assessment of alternative routes for a highway development in northern Italy.

2. Methodology

The methodological approach can be divided into three main steps: ecosystem mapping, ecosystem evaluation, and ecosystem-loss impact assessment.

2.1. Ecosystem mapping

The natural ecosystems occurring within the area affected by the road project under consideration are to be identified and mapped, so as to provide the basic data layer to perform the impact assessment. Natural ecosystems are biotic communities that could spontaneously occur in a given abiotic environment. They are characterised by the presence of an essentially intact, at least in its main features, native vegetation cover. On the contrary, artificial ecosystems (e.g., urban settlements, agricultural fields) or semi-natural ecosystems (e.g., pastures) are originated by human activity and their stability depends (to different degrees) upon human interventions. Biodiversity conservation aims at maintaining ecologically functional examples of each type of naturally occurring ecosystem in a region (Noss and Cooperrider, 1994; CEQ, 1993). For this reason, the BIA proposed in this study focuses on the impacts on natural ecosystems only.

A common shortcoming in EIA is to narrow the biodiversity considerations to the sites and ecosystem types that have been already designated for nature conservation (nature reserves, biotopes, Sites of Special Scientific Interest, priority habitats, etc.). In particular, recent reviews testify how the BIA of road developments is being perceived mostly as analysis of the effects on selected protected areas or species (Geneletti, 2002; Byron et al., 2000; Seiler and Eriksson, 1995). This represents a very severe limitation with potentially harmful consequences for the conservation of biodiversity. As a matter of fact, the designated sites often cover small areas and may not comprise the whole home range of some of the species. Furthermore, a number of habitats and associated features with a relatively high nature-conservation value are likely to occur that should be investigated carefully, even if not designated as such. Quoting Treweek et al. (1998): "there is an unquantified risk to habitats and species that do not, individually, merit designation or protection but which nevertheless can constitute an important part of a country's natural capital." A more comprehensive approach than just focusing on designated areas has been envisaged, for example, by the USA guidelines on incorporation of biodiversity into EIA (CEQ, 1993), as well as by the UK manual for the environmental assessment of road schemes (DoT, 1993). For these reasons, it was decided to include in the assessment all the natural ecosystems occurring within the area affected by the project.

By far, the most common method for mapping ecosystems consists in mapping the vegetation types. According to this approach, vegetation communities are considered as representative for delimiting the boundaries of ecosystem units. This assumption is justified by the fact that vegetation communities typically show a strong relationship with both their physical environment (soil and rock type, climate, topography, etc.) and the organisms they host. Furthermore, vegetation mapping represents a feasible alternative to carrying out a truly complete biological survey. For these reasons, it is widely held that vegetation types can be used as surrogates for the ecosystems in which they participate (Csuti and Kiester, 1996; Noss and Cooperrider, 1994; Spellemberg, 1992; Austin and Margules, 1986). Consequently, a map showing the distribution of the vegetation cover within the area can be considered as a suitable ecosystem map. Such a map must have an adequate spatial resolution and date. As to the spatial resolution, it has to be compatible with the size of the project. We must be able to clearly locate the road and predict its effects on the surrounding ecosystems. According to the type of roads, this usually implies a mapping scale ranging from 1:5000 to 1:25,000. As for the date, the map obviously should depict the landscape in the closest-to-present possible conditions.

2.2. Ecosystem evaluation

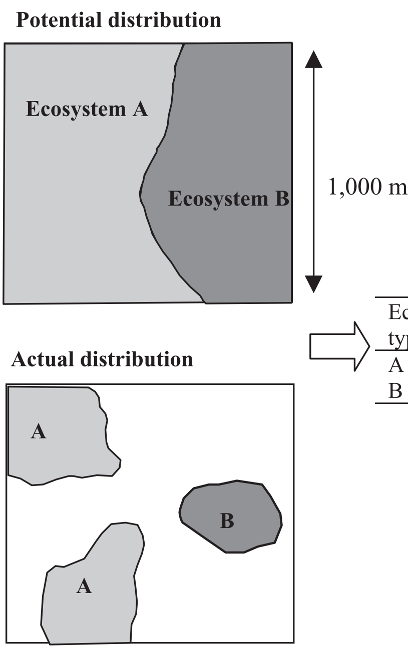
Rarity is proposed as a criterion to evaluate the relevance of the different ecosystems in terms of biodiversity conservation. In its broadest definition, rarity refers to how frequently an ecosystem type is found within a given area. Biodiversity conservation aims at preserving the full richness of life on earth. Consequently, it appears logical that the actual cover and distribution of an ecosystem type influence its relevance and protection worthiness. The rationale behind the use of the rarity criterion is the consideration that the rarer is a feature, the higher is its probability of disappearance.

Despite the broadly shared vision about the relevance of the rarity criterion, there is not to be a common ground for its evaluation. Rarity is a relative term and it has been used in a variety of ways in ecological evaluations (Smith and Theberge, 1986). This has lead to the use of a number of indicators, based on measurable scales (e.g., the relative abundance of the feature within a region, as in Pressey and Nicholls, 1989), or more commonly, on personal perceptions and linguistic scales (e.g., by using expressions such as "rarely found" or "commonly found", as in Andreis, 1996 and Wittig et al., 1983). The indicators based on measurable scales are more appealing for impact studies because they offer an objective basis for the assessment. However, the use of measurable indicators to express rarity requires the setting-up of an appropriate reference system.

The ideal goal of biodiversity conservation is to maintain pre-settlement ecosystem types in approximate proportion to their former abundance in the region (Noss, 1983; Southerland, 1995). That is, the reconstruction of the past situation is to provide a sound basis for addressing biodiversity protection policies, and consequently for assessing ecosystem significance. Therefore, the past situation appears as the most suitable reference, against which to assess losses and quantify rarity. Consequently, the rarity of an ecosystem type within a region can be expressed by using as indicator the percentage of its original area remaining. This approach is envisaged by current strategies for biodiversity planning, such as "GAP analysis" (Scott et al., 1996), in which the relevance of accounting for the pre-settlement extent of the different vegetation types is stressed. Obviously, the main problem in applying this method relates to the reconstruction of the pre-settlement landscape.

The most common tool for depicting the original spatial distribution of the different ecosystems within a landscape is represented by a potential-vegetation map. Such a map sketches the vegetation types that potentially could grow in a given area on the basis of its climate, soil, water conditions, lithology and topography. Potential-vegetation maps are to represent the features of a landscape as they were before human disturbances and interventions took place. The use of indications provided by the potential vegetation for assessing the biodiversity value of natural areas is mentioned in several studies (Stoms, 2000; Awimbo et al., 1996; Fandiño, 1996; Noss, 1983; Wright, 1977). However, this has not lead to the proposal of a quantitative indicator. The studies only suggest the use of information on the potential and actual ecosystem distribution for: prioritising areas for nature conservation (Stoms, 2000; Awimbo et al., 1996; Wright, 1977); providing a reference to set the objectives of biodiversity preservation activities (Noss, 1983); assessing the ecological quality of vegetation communities (Fandiño, 1996).

The use of the ratio between the actual cover and the potential cover of each natural ecosystem type is proposed here as indicator to express ecosystem's rarity and it is termed Potential Area Remaining (PAR). Fig. 1 shows an example of its computation. As it can be seen, the required input consists of two maps showing respectively the potential and the actual distribution of the ecosystems within the area of interest. The advantage of using this indicator resides in the fact that the selected criterion (i.e., rarity) can be measured for each ecosystem type in an objective and replicable way. Furthermore, the reference against which rarity is measured, i.e., the potential distribution of the ecosystems, is clearly stated and provided, overcoming the limits of "black-box" approaches. However, rarity can be meaningfully described only by referring to a scale of analysis (local, regional, etc.). Therefore, it is necessary to specify the reference area within which rarity is to be measured. An ecosystem type can have, for instance, a high local rarity, but not so high a regional rarity. The reference area has to be consistent with the overall scale of analysis of the EIA, and if possible with existing policies for biodiversity conservation.



Ecosystem type	Potential area [ha]	Actual area [ha]	PAR [%]
A	60	30	50
B	40	8	20

Fig. 1. Examples of computation of the rarity indicator for two ecosystem types (A and B). PAR stands for Potential Area Remaining.

Once the rarity has been measured for each type of ecosystem in terms of percentage of Potential Area Remaining, the next step is the actual assessment of such percentages. That is, a relationship needs to be established between the indicator measurements and the perceived ecosystem values. In terms of protection worthiness for biodiversity conservation, rarer ecosystem types should be ranked higher than common ones. Therefore, the highest value scores are to be assigned to the ecosystems that have experienced the largest decline with respect to their original area.

The problem resides in finding a suitable value score for each measured value of the indicator, i.e., drawing a functional curve (Dee et al., 1972), as in the example of Fig. 2. This implies answering questions such as "how relevant is an ecosystem type that experienced a loss of *X*% with respect to one that experienced a loss of *Y*%?" Because there appear not to be an objective basis for carrying out such an assessment, the evaluation is to rely on existing targets set in policies for biodiversity conservation or on expert's judgements. Regardless of the difficulty of making such judgements, they are an inherent part of the evaluation, and must be properly addressed during the impact assessment.

Fig. 2 shows a hypothetical functional curve for the rarity indicator. The curve is monotonically decreasing, because the lowest indicator values correspond to the ecosystem types that experienced the largest decline, hence that are considered more valuable. In the example, a score of one was assigned to indicator values smaller than 10. This means that the ecosystem types whose actual cover is less than 10% of their original one are assigned the maximum relevance, with respect to the objective of preserving biodiversity in the study area.

The application of the functional curve allows to attach to each ecosystem a value score, which is to be used during the impact assessment to estimate the relevance of the expected ecosystem losses. The approach offers the advantage of

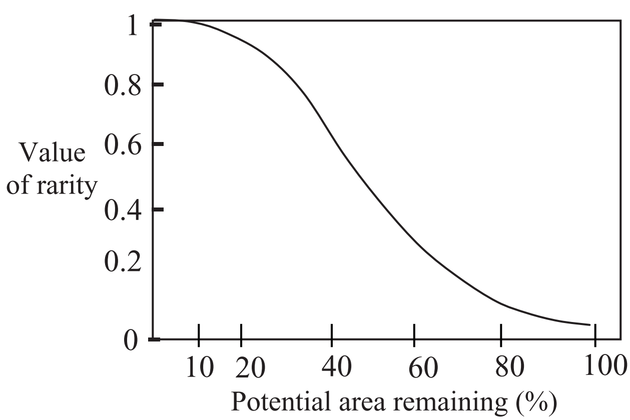


Fig. 2. Example of functional curve for rarity.

relying on an objective and quantitative indicator to express rarity, and of introducing subjective judgments only in the last stage of the evaluation. Such judgments are expressed in a transparent way so that the relationship between the facts (i.e., the Potential Area Remaining of each ecosystem type) and the values (i.e., the level of importance attached to each percentage) is explicitly provided. Subjectivity is an inherent component of each evaluation and cannot be eliminated. However, as maintained by Antunes et al. (2001), the results of an evaluation may become more credible if they are obtained by the application of an a priori-defined and transparent methodology, with clearly stated assessment criteria and making full use of the available information.

It is worth noting that the proposed indicator does not account for the inherent rarity of ecosystems, but only for the rarity that resulted from human interventions. Therefore, ecosystems that are rarely found in nature (e.g., because they require specific environmental conditions or because they are particularly fragile), but that are currently well preserved will get a low score. This is because the approach, as further discussed later on in the text, is tailored to ecological assessment in man-dominated landscapes. Such landscapes are usually devoid of inherently rare and fragile ecosystems, and therefore require specific criteria and indicators to distinguish different degrees of protection worthiness. The proposed indicator allows doing that, starting from the fundamental objective of biodiversity conservation, which is to maintain the landscape features as similar as possible to the original ones. Concluding, the use of the rarity indicator to assess ecological relevance is certainly not enough to carry out complete ecological evaluations, but it appears suitable for the need of a BIA within urbanised landscapes.

2.3. Assessing the ecosystem-loss impact

The total amount of land that is to be occupied by the completed road scheme is defined as "land-take" (Byron et al., 2000; Treweek et al., 1993). A common approach to estimate the ecosystem-loss impact consists in representing the land-take through a buffer that extends along the road axis (De Amicis et al., 1999; Treweek and Veitch, 1996; Patrono, 1993; Sankoh et al., 1993). Ecosystems lying within the buffer are considered to be lost. The width of such a buffer ranges from few tens to few hundreds meters, according to the view of different authors and to the road type (see, for example, Stoms, 2000; Heinrich and Hergt, 1996; Lanzavecchia, 1986).

Following this approach, the input required to predict the ecosystem-loss impact is represented by a map of the natural ecosystems occurring within the study area and by a map of the alternative road layouts proposed for the project under consideration. First of all, it is necessary to estimate a space-occupation buffer for each of the alternatives. Afterwards, the ecosystem loss can be computed by overlaying such a buffer with the ecosystem map. The overall impact is quantified by multiplying the value of each ecosystem type (computed as presented in the previous sub-section) for its predicted area loss and by summing-up the result. Consequently, for each project alternative, the impact score is to be assigned according to the following expression:

$$\mathrm{EL}_i = \sum_{j=1}^{n} (A_j V_j) \tag{1}$$

where: EL_i = ecosystem-loss impact score of alternative i; A_j = predicted area loss for ecosystem type j; V_j = assessed rarity value of ecosystem type j; n = number of ecosystem types.

The computation of the impact score allows to compare the project alternatives and rank them according to their expected impact in terms of natural ecosystem loss.

The remaining of the paper illustrates and discusses the application of the methodology to a case study.

3. Application to a case study

3.1. Project and study area

The study area includes a stretch of the Adige Valley, a large alpine valley that crosses the central part of the Autonomous Province of Trento, in northern Italy (see Fig. 3). Despite the high density of infrastructures, this sector of the valley suffers from traffic congestion. To reduce the pressure of traffic, the local authorities have decided to strengthen the existing road network by constructing a new motorway. The motorway is to provide a connection between the northern outskirt of the town of Trento and the beginning of the Non Valley (see Fig. 3). Six routes proposed to link these two locations will be considered in this study, as shown in Fig. 4. They have a length of about 20 km and they all include at least one tunnel to cope with the complex geomorphology and with the intensive land use that characterise the area.

The area crossed by the road development is a typical man-dominated landscape in which few natural ecosystems remain within an artificial matrix, made of urban settlements and cultivated fields. The fertile and flat soils of the alluvial plain and fans offer excellent conditions for agriculture. That is reflected by the intensive use of the land for products that have a high economic value, such as vineyards and apple orchards. The scattered remnants of natural vegetation are represented mainly by wetlands and by patches of meso-phytic woodlands. Owing to their scarceness within the Adige Valley and the whole Province, these remnant ecosystems play a relevant role for the conservation of biodiversity, providing a suitable habitat for several rare and endangered species and an important stepping stone along the migratory routes. The study area is crossed from North to South by two rivers: the Noce and the Adige (see Fig. 3).

The design of the road alternatives reveals the intention of sparing the valuable agriculture land. As a result, the selected routes appear to “connect the dots” among the remnant natural areas, posing a serious threat to their conservation.

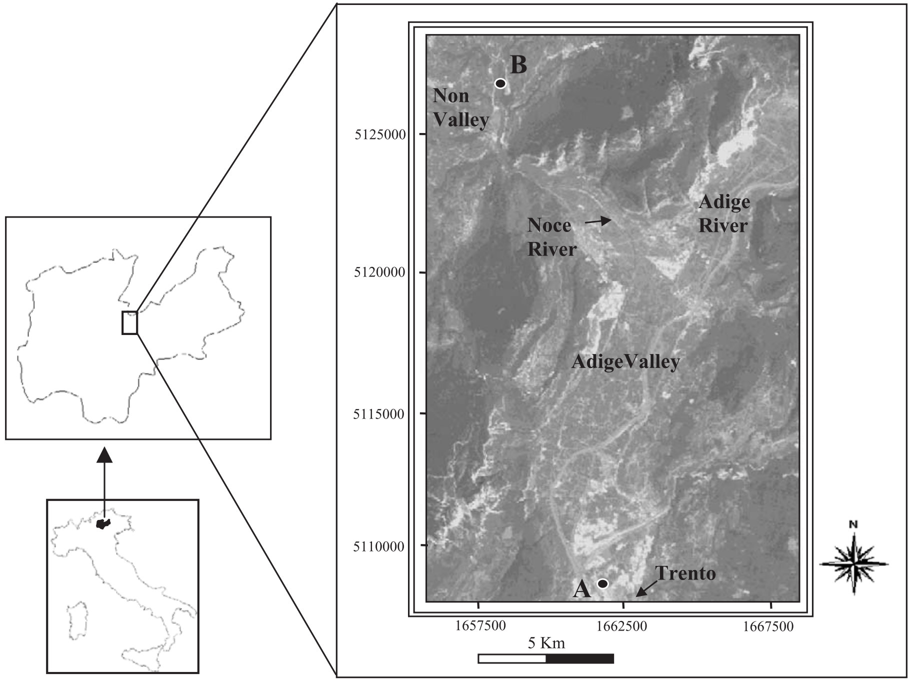


Fig. 3. Location of the Autonomous Province of Trento in Italy and of the study area within that Province. A and B indicate the locations to be connected by the road project.

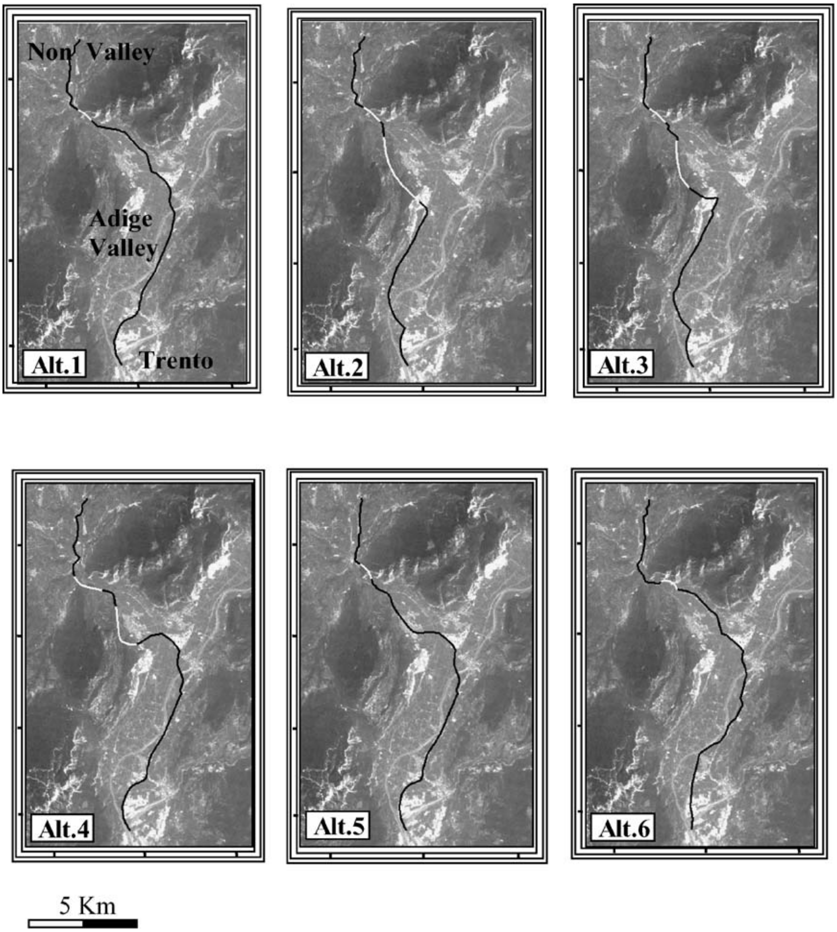


Fig. 4. The six road layouts (tunnels are represented in white).

For ecological concerns to be actually included in the decision-making and balanced against the other effects of the road, it is fundamental to carry out a sound appraisal of the expected impacts on biodiversity. In particular, a clear evaluation of the ecological relevance of the land along the different routes and a subsequent assessment of the value loss caused by the project need to be provided. This is presented in the remaining of the paper. This study is part of a broader research project in which the impact of the proposed road alternatives has been assessed with respect to several environmental and socio-economic criteria, such as geomorphologic hazards, agriculture productivity, and visibility (Geneletti, 2000; Alkema et al., 2000).

3.2. Ecosystem mapping

The natural ecosystems remaining within the Adige Valley's floors were mapped at a 1:10,000 scale by integrating the woodland maps available at the Forest Survey of the Autonomous Province of Trento with the interpretation of a set of colour aerial photos, supported by extensive field surveys (Geneletti, in press). The photos were acquired in July 2000 and have a spatial resolution of 2 m. The different ecosystem types were classified according to the dominant or co-dominant species in the uppermost vegetation layers, as suggested by Csuti and Kiester (1996). The resulting ecosystem map is shown in Fig. 5. Five different types of natural ecosystems were found in the study area (the phytosociological information has been derived from Pedrotti, 1981, 1982):

- Beech woodland. It is characterised by the dominance of European beech (*Fagus sylvatica*). From a phytosociological viewpoint, it has been described

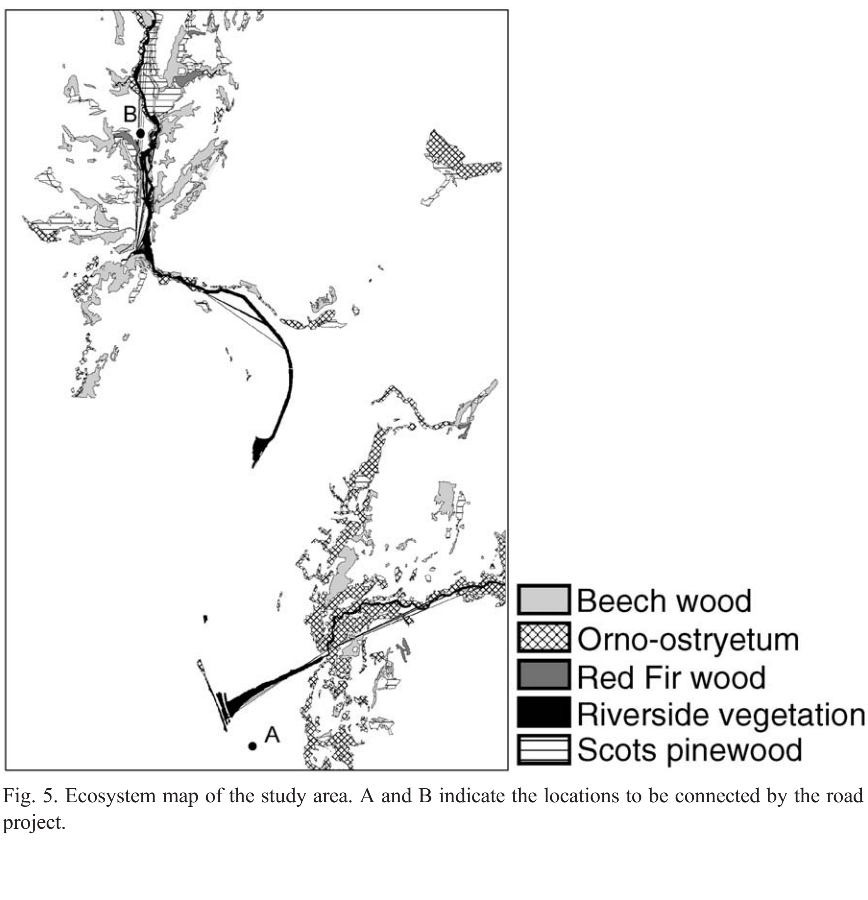


Fig. 5. Ecosystem map of the study area. A and B indicate the locations to be connected by the road project.

as *Carici-fagetum* and *Luzulo-fagetum*. It is present especially within the Non-Valley.

- *Orno-ostryetum* woodland. It is characterised by the co-dominance of two tree species, hornbeam (*Ostrya carpinifolia*) and ash (*Fraxinus ornus*), and by the presence of oak (*Quercus pubescens*). Owing to its peculiar tree composition, it was decided to refer to this woodland type by using its phytosociological name, which makes it immediately recognizable. It is the most common (in terms of surface) natural vegetation type that can be found within the study area.
- Red Fir woodland. It is characterised by the dominance of the Red fir (*Picea excelsa*) and by the presence of larch (*Larix decidua*), and more rarely *Pinus Cembra*. From a phytosociological viewpoint, it has been described as *Picetum montanum*. It is present in few small patches mainly scattered within the Non-Valley.
- Scots pinewood. It is characterised by the dominance of the Scots pine (*Pinus sylvestris*) and it belongs to the phytosociological association *Erico-Pinetum sylvestris*.
- Riverside vegetation (Rv). It is characterised by the presence of Black alder (*Alnus glutinosa*), Speckled alder (*Alnus incana*), and more rarely willow (*Salix alba* and *Salix elaeagnos*). It is made of several phytosociological association, among which the most common is the *Alnetum glutinoso-incanae*. It is present along few stretches of the main rivers that cross the area.

3.3. Ecosystem evaluation

The first step of the ecosystem evaluation consisted in computing the PAR indicator for each ecosystem type. As discussed in Section 2.2, assessing the rarity of the ecosystems requires the identification of a reference area, within which to compute the percentages of the potential cover remaining (see Fig. 1). A meaningful reference area for the case study is represented by the entire territory of the Autonomous Province of Trento (APT). This is because such an area represents also the administrative context within which land-management decisions are taken. However, a suitable ecosystem map of the whole province is not available, and constructing it from scratch is too big an effort for the present study. Consequently, a smaller reference area was selected by extending the study area to the surrounding mountains and valleys, so as to include roughly the whole rectangular area pictured in Fig. 3. This area comprises a wide range of landscapes, and to some extent, it can be considered as a representative sample of the territory of the APT.

A potential-vegetation map was available for the whole reference area (Pedrotti, 1982, 1981) and was digitised in the GIS database generated for this research. This allowed comparing it with the actual ecosystem map. In particular, the areal coverage of the five ecosystem types was computed for both maps and used to calculate the PAR indicator values (see Table 1).

Table 1
Computation of the rarity indicator for the relevant ecosystem types

Ecosystem type	Potential extent (ha)	Actual extent (ha)	Rarity (% PAR)
Scots pinewood	2574	2471	96
Red fir wood	1139	1048	92
Orno-ostryetum	6945	4653	67
Beech wood	5249	1522	29
Riverside vegetation	4742	142	3

As expected, the riverside type of vegetation is the one that experienced the greatest loss. Prior to human settlements, this ecosystem type used to occupy most of the valley floors, but it is now limited to a narrow strip along the main rivers and to few isolated patches. Also, the meso-phytic broadleaf woodlands, such as the ones dominated by beech, hornbeam and ash trees, paid their toll to the human interventions in the area, and especially to the cultivation of vineyards and orchards. Such crops have replaced the original expanses of forest within the most favourably oriented slopes.

The next step of the procedure consisted in the actual assessment of the measured indicator values. This means that these values need to be transformed into a degree of relevance with respect to the achievement of the general objective of the evaluation, i.e., the preservation of the natural biodiversity. Such a degree of relevance can be expressed by a value score, ranging between zero and one. One corresponds to the highest relevance, i.e., to ecosystems whose remnant cover has dramatically decreased, posing a serious threat to their chances of conservation within the area. Zero corresponds to the lowest relevance, i.e., to ecosystems whose original cover is virtually entirely preserved within the area. The construction of a functional curve makes explicit the relationship between the measured indicator values and their perceived relevance. This allows comparing the relative value of the different ecosystem types, and consequently quantifying the impacts caused by their loss.

In this research, the author provided the expert's opinion required to construct the functional curve. The main purpose here was to illustrate the applicability of the proposed approach. Ideally, the value assessment, and therefore the construction of the functional curve, should be carried out by selecting a suitable panel of experts (possibly including local authorities and public administration specialists) and by attempting to reach consensual opinions (e.g., through a Delphi survey, see, for instance, Hess and King, 2002).

For simplicity, a linear piece-wise functional curve was adopted in this study (see Fig. 6). As shown by the curve, ecosystems whose actual cover is less than the 15% of their original cover were given the maximum relevance for biodiversity conservation (i.e., the value of 1). On the other hand, the curve never reaches the zero value. In particular, ecosystems with at least the 90% of their original cover were assigned a value of 0.15. This is because, even if an

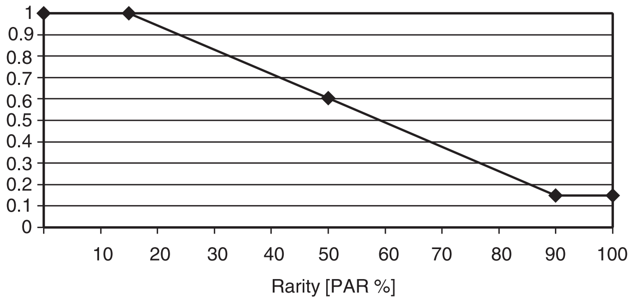


Fig. 6. The functional curve for rarity.

ecosystem type fully preserved its original cover, it was still considered to play a role in biodiversity conservation, because only natural ecosystems are included in the evaluation. Conversely, assigning to it a zero value would have been equivalent to state that its loss did not represent an impact.

It is worth remembering that the functional curve expresses relative differences in the value of ecosystem types characterised by different rarity levels. So, for instance, an ecosystem with a PAR of 100% was considered almost seven times less valuable than an ecosystem with a PAR of 5% (see Fig. 6). The functional curve was used to compute the values of the five ecosystem types found within the study area (see Table 2).

3.4. Ecosystem-loss impact assessment

The first step in predicting the impacts of the space occupation of the project alternatives consists in estimating the extent of such an occupation. Operationally, this implies identifying and mapping the land-take that is expected, according to the technical characteristics of the road project. The road project analysed in this research is to become mainly a four-lane highway. For similar types of roads, the occupation buffer more often proposed in the literature is of about 120 m, i.e., 60 m from each side of the road (Heinrich and Hergt, 1996; Patrono, 1993;

Table 2
Value assessment for the relevant ecosystem types

Ecosystem type	Rarity (% PAR)	Assessed value (V)
Scots pinewood	96	0.15
Red fir wood	92	0.15
Orno-ostryetum	67	0.41
Beech wood	29	0.84
Riverside vegetation	3	1

Lanzavecchia, 1986). Therefore, it was decided to use such a value. A 120-m buffer was computed for each of the project alternative. The buffers extend only from the open tracks of the alignments, because the tunnel sections are not to affect the surface. On the other hand, tunnels cause other significant environmental impacts (transportation and disposal of quarry waste, noise and air pollution during the construction phase, etc.), and therefore in practice, they represent an extremely important factor in the overall assessment of the alternatives. However, such impacts are not dealt with in this research.

The buffer maps of the different project alternatives were overlaid with the map showing the distribution of the natural ecosystems. This allowed to compute the expected loss for each ecosystem type. Such losses are presented in Table 3. As it can be seen, the riverbank vegetation represents the habitat type that is to suffer the most from the presence of the new infrastructure. This is mainly due to the fact that all the six alternatives heavily interfere with the fluvial system of the two main rivers, by running along them and crossing them. The design of the road alignments revealed the willingness of sparing the highly valuable agriculture land at the river corridor's expenses.

The overall ecosystem-loss impact is quantified by multiplying the assessed rarity value of each ecosystem type for its predicted area loss in the corresponding alternative and by summing the results, according to expression (1). As a result, each project alternative is assigned a synthetic impact score, as shown in Table 3. The units of the impact scores were defined as "weighted hectares" because they represent the losses in hectares weighted by the relevance (i.e., the assessed rarity value) of each ecosystem type affected.

As discussed in the previous sub-section, the computation of the impact scores relied on the assumption that the losses of different ecosystem types are interchangeable, according to the value system expressed by the curve shown in Fig. 6. This implies, for example, that losing 1 ha of an ecosystem with a value of 0.2 is equivalent to losing 2 ha of an ecosystem with a value of 0.1.

Table 3
Computation of the ecosystem-loss impact scores (EL_i)

	Ecosystem losses										EL_i [weighted ha]
	Scots pinewood		Red fir wood		*Orno-ostryetum*		Beech wood		Riverside vegetation		
Alternative 1	0.31	0.15	0.00	0.15	6.86	0.41	12.93	0.84	33.69	1.00	47.41
Alternative 2	0.85	0.15	0.00	0.15	3.94	0.41	12.45	0.84	26.79	1.00	38.99
Alternative 3	2.05	0.15	0.00	0.15	10.62	0.41	6.34	0.84	27.42	1.00	37.40
Alternative 4	1.74	0.15	0.00	0.15	3.78	0.41	15.04	0.84	23.07	1.00	37.51
Alternative 5	0.31	0.15	0.00	0.15	4.69	0.35	13.91	0.84	16.41	1.00	29.78
Alternative 6	0.42	0.15	0.00	0.15	12.11	0.41	8.02	0.84	35.74	1.00	47.50

For each ecosystem type, the predicted area loss in hectares (left column) and the assessed value (right column) are indicated.

Consequently, the aggregated impact scores have a clear interpretation: a score of 10 means that the alternative causes the loss of 10 ha of an ecosystem with a value of one (i.e., an ecosystem whose PAR is less than 15%, see Fig. 6) or any possible equivalent combination of ecosystem losses (e.g., the loss of 20 ha of an ecosystem with a value of 0.5, i.e., an ecosystem whose PAR is of about 60%).

The ecosystem-loss impact assessment illustrated in this section shows that Alternative 5 is the best-performing one (see Table 3). As it can be seen by comparing Figs. 4 and 5, the alignment of Alternative 5 spares most of the remnant natural areas by crossing the eastern side of the Adige Valley and by making use of tunnels or of existing infrastructure corridors. Alternatives 2, 3, and 4 perform similarly, causing a loss of slightly less than 40 weighted hectares. Finally, the alignments of Alternatives 1 and 6 appear the least suitable in terms of ecosystem preservation. The reason of the intense ecosystem loss caused by these alternatives resides in the fact that they follow long stretches of the Noce river, affecting the highly valuable vegetation that cover its banks.

The analyses performed in this section provided a numerical estimation of the impact on natural ecosystems caused by the proposed alternatives. This result represents a piece of the mosaic of the whole EIA and it is to be integrated with the assessment of further types of impact on biodiversity (i.e., fragmentation), as well as on the other environmental components. This is to provide the decision-makers with as clear a picture as possible of the trade-offs related to impacts of the different alternatives, so as to guide their choice.

4. Discussion and conclusions

Transportation infrastructures, and above all road networks, are blamed for highly contributing to the decrease in both the quantity and the quality of natural habitat, posing a threat for the conservation of biodiversity. Therefore, a sound BIA in road planning and development needs to be coupled to other commonly considered elements, such as traffic prediction or geomorphologic risk assessment.

This paper presented and applied a methodological approach to perform BIA of roads, focusing in particular on the impact caused by the direct ecosystem loss. The approach is not meant to provide a comprehensive guidance to BIA, rather to propose some fundamental analyses that can help in making BIA as routinely undertaken within the EIA, as other forms of impact assessment (noise, air pollution, etc.). The ultimate objective of the research is to encourage good practice within EIAs, so as to eventually strengthen the consideration of ecological issues in the decision-making for new developments.

The remainder of the section contains a discussion of the advantages and the limitations related to the application of the proposed approach that emerged during the analysis. A particular attention has been given to issues concerning its operationalisation, and therefore its potential role in real-life EIAs.

4.1. Biodiversity levels

A thorough study on biodiversity should include the analysis of all the four levels in which biodiversity is typically subdivided, namely the genetic level, the species level, the ecosystem level, and the landscape level (CEQ, 1993). For this reason, a preliminary scoping should be carried out to identify the biodiversity levels that are likely to be affected by the project under analysis. Addressing biodiversity in all its complexity is most of the time not required. Landscape diversity, for instance, is to be considered only when dealing with projects crossing different types of landscape, and therefore affecting their spatial variability and patterning.

The methodology proposed in this paper focuses on the ecosystem level only. This level is usually the most relevant, and consequently the one that need to be fully investigated when dealing with road projects. Moreover, as stated in Noss and Cooperrider (1994), conservation is in many cases most efficient when focused directly on ecosystems: by maintaining intact an ecosystem, the species that live in it will also persist. Following this principle, recent nature-conservation strategies have partially shifted from single-species management to the so-called "coarse-filter" approach (Stoms, 2000; Noss, 1987). The objective of the coarse-filter approach is to protect most species in a region by conserving samples of every ecosystem type.

However, addressing the ecosystem-level only during an EIA may not be enough. In some cases, it may be necessary to undertake specific studies on the genetic or species level too. For instance, different ecosystems could be valued not only for the rarity of their vegetation cover, but also by virtue of the presence of rare or endangered or, anyway, relevant animal and plant species. Unfortunately, such analyses require a considerable amount of information on the distribution of species with a level of spatial detail that is usually unavailable. Therefore, additional surveys are often required. Such surveys are extremely resource-consuming, and should be limited to the areas where they are actually needed. In similar situations, the impact assessment performed in this study could represent a preliminary analysis aimed at highlighting those sites worth of further investigations.

4.2. BIA and man-dominated landscapes

The proposed approach is particularly tailored for applications in man-dominated landscapes for the following reasons:

- It is based on the systematic assessment of the impact on all the natural ecosystems present in the study area. This avoids the common shortcoming of addressing only sites already designated for nature conservation, which are typically missing in man-dominated landscapes.

- It is based on the use of current and potential maps of the vegetation cover, therefore on data relatively common or anyway simple to acquire. Further ecological data are usually not available in urbanised landscapes.

Due to the scarceness of unspoilt areas, there is a growing interest, especially in Europe, on the conservation of biodiversity within man-dominated landscapes. This is testified by recent efforts in planning and establishing ecological networks within agricultural and urbanised areas (Geneletti and Pistocchi, 2001; Cook, 2000; Jongman, 1995). Consequently, a sound BIA must be carried out even for projects affecting areas that have already experienced a heavy human pressure. This calls for an adequate evaluation of the remnant natural ecosystems, as proposed in the methodology applied in this paper. If this is not performed, such areas face the risk of being quickly labeled as "devoid of any ecological relevance", opening the way to uncontrolled impacting activities.

4.3. Road planning and EIA

The assessment of the impacts caused by the different alternatives was performed by referring to a general space-occupation buffer, estimated from the available information about the project. Such information was limited because the project was only in the planning phase. In particular, most technical details, such as the exact width of the paved area, the presence of bridges or viaducts, the location of the road-yard and parking lots, were missing. Such details are fundamental to permit a more reliable quantification of the space occupation of the infrastructure. On the other hand, developers are reluctant to finance the design of operational projects when several alternatives are proposed and the final one has not been identified yet. Therefore, this case study can be considered as a likely situation.

EIA studies dealing with several alternatives need to expect not be able to count on final blueprints, but rather on designs that are not fully detailed. Even though based on imprecise information, the EIA has the task of identifying the most suitable land corridor through which the project can be designed. Once the least-impacting corridor has been identified, the appointed engineers provide the operational project. At this point, the impact assessment can be refined. This is particularly relevant when the law enforces compensation measures for impacts such as habitat loss (see, for instance, Cuperus et al., 2001, 1999). Adequate compensations can be proposed only when the project details are known, and consequently the losses can be estimated more precisely.

Concluding, the approach proposed for impact assessment is suitable as a preliminary way to identify and suggest the preferred alternative layout. To estimate more precise impact figures, and consequently to propose adequate compensation measures, the detailed operational project of the selected alternative is required.

4.4. Ecosystem evaluation

The approach proposed relied on the evaluation of the significance of the different ecosystem types in terms of biodiversity conservation. This required the establishment of a "value system" (i.e., the functional curve shown in Fig. 6). However, it appears neither advisable nor feasible that each and every EIA generate its own value system for the assessment of rarity, or whatever other ecological criterion. This would result in a very heterogeneous assessment for the same object throughout the same region, and would complicate the procedure of both compiling the EIA report and reviewing it. It appears much more adequate, considering both feasibility and effectiveness, that the local administration provides its own standards and its own value systems in the form, for instance, of a Biodiversity Conservation Plan, with which the evaluation performed in every EIA must comply.

Acknowledgements

This research activity was supported by the Training and Mobility of Researchers Network Programme (TMR) of the European Union (GETS contract no. ERBFMRXCT970162). I would like to thank Prof. A.G. Fabbri, Prof. H.J. Scholten, and Dr. E. Beinat of the Free University of Amsterdam for their helpful suggestions.

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