Ecological scoping

Issues and dilemmas in ecological scoping: scientific, procedural and economic perspectives

Yael Mandelik, Tamar Dayan and Eran Feitelson

Prior research has shown that ecological scoping is the most important factor in determining the quality of environmental impact statements in Israel. Hence, improved ecological scoping has been called for. This paper identifies and discusses four fundamental dilemmas that need to be addressed in the ecological scoping process: biodiversity assessments, appropriate spatial and temporal scales, and cumulative ecological effects. The scientific, procedural and economic aspects of these dilemmas are discussed and practical suggestions for scoping presented: a modular ecological scoping process for early identification of the most detrimental projects, and a generic blueprint for ecological scoping, to help craft case-specific ecological guidelines.

Keywords: biodiversity; cumulative effects; ecological impact assessment; ecosystem perspective; Israel; scoping

At the time of writing, Yael Mandelik was a PhD student, Department of Zoology, Tel-Aviv University, Israel. She is now in Department of Ecology and Evolutionary Biology, Princeton University, Princeton, NJ 08544 USA; Tel: +1 609 258 7891; Fax: +1 609 258 7892; E-mail: yaelm@princeton.edu. Tamar Dayan is associate professor, Department of Zoology and head, Institute of Nature Conservation Research, Tel Aviv University, Tel-Aviv 69978, Israel; Tel: +972 3 6409024; Fax +972 3 6409403; E-mail: dayant@post.tau.ac.il. Eran Feitelson is associate professor, Department of Geography and head, School of Public Policy, The Hebrew University of Jerusalem, Israel; Tel: +972 2 5883346; Fax: +972 2 5820549; E-mail: msfeitel@mscc.huji.ac.il.

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BIODIVERSITY WORLDWIDE is threatened by a host of human-induced factors, first and foremost of which are habitat loss and fragmentation (Pimm and Raven, 2000). In terrestrial ecosystems, rapid development, particularly in heavily populated countries, is the major source of these effects (Terborgh, 1999). Planning procedures are thus a major conservation tool on a variety of geographical scales (Sutherland, 2000). In recent years, considerable significance has been attached to developing local-scale planning tools, the most widely used of which is environmental impact assessment (EIA).

EIA, first established in the USA in 1969, has spread worldwide and is now formally practiced in over 100 countries (Glasson *et al*, 1999). EIA has become a prominent tool of environmental planning and management (Morgan, 1998). In many instances, it is the only stage in the planning process in which ecological consequences of local development actions are being considered.

Article 14 of the Convention on Biological Diversity (CBD) explicitly addresses impact assessment when requiring parties to apply EIA to projects with potential adverse effects on biodiversity, singling it out as a potential implementation tool (Slootweg and Kolhoff, 2001). Both the CBD and the Ramsar Convention have adopted guidance on addressing biodiversity issues in EIA (Treweek, 1999). In addition, international lending agencies, such as the World Bank, require EIA for projects put forward for funding, so use of this planning tool is likely to increase (Morgan, 1998). Hence, it has a prominent role in promoting conservation and sustainable use of biodiversity on both local and wider scales.

However, studies of the quality of the ecological component of EIAs have revealed very low standards throughout the process, including baseline descriptions, impact assessments and evaluations, and mitigation measures advanced. An overview of research papers on the ecological input in EIA demonstrates remarkably similar findings in almost all EIA systems investigated to date (Treweek *et al*, 1993; Treweek, 1996; Thompson *et al*, 1997; Warnken and Buckley, 1998; Gray and Edwards-Jones, 1999; Atkinson *et al*, 2000; Byron *et al*, 2000; Tanaka, 2001; Mandelik *et al*, 2005; UNEP/CBD/SBSTTA, 2003 and references therein).

A summary of the main findings of these studies is listed in Table 1. Noteworthy is the almost unanimous failure to quantify and evaluate ecological impacts comprehensively, reluctance to address complex, cumulative and indirect effects, and reluctance to apply broad conceptual and spatial scales. Hence, while EIA has a vital role to play in the conservation and integration of biodiversity aspects in planning procedures, there is an urgent need to improve its scientific quality.

Previous work has shown that the effectiveness of the EIA process depends primarily on the quality of scoping, that is, the stage where the issues, scales and methods to be addressed are determined (Kennedy and Ross, 1992). Moreover, prior research the Israeli EIA system has identified the guidelines set at the scoping stage as the most significant factor in determining the quality of the ecological components of environmental impact statements (EISs) (Mandelik *et al*, 2005). Hence, an improvement in ecological scoping is called for.

Naturally, as EIA systems reflect the societal and political norms in the planning arena, a wide range of EIA and scoping systems is found in different countries. Scoping systems can be classified according to two orthogonal aspects (Haklay *et al*, 1998): the way the issues to be addressed are determined (based on either expert opinion or participatory approach, that is, stakeholders' input), and who has the leading role in the process (whether a regulatory authority or the proponent).

Table 1. A summary of major shortcomings identified in ecological impact assessment of different EIA systems

Baseline description:

- Failure to address appropriate spatial scales
- Failure to address all components of biodiversity
- Lack of quantitative data
- Low standards of field surveys (reluctance to address spatial and temporal variation)

Impact prediction:

- Omitting key impacts
- Reluctance to quantify impacts
- Reluctance to evaluate the significance of impacts
- Failure to address cumulative, indirect and complex effects

Mitigation and monitoring:

- Severe impacts left un-mitigated
- Recommendation of un-testable measures
- Reluctance to evaluate the efficacy of proposed measures
- Reluctance to mention the need for or propose adequate monitoring program

Interestingly, different scoping systems produce highly similar EISs from the ecological perspective. For example, although the UK, USA, and Israel have distinctively different scoping systems (Brachya and Marinov, 1995), they strongly resembled each other in many review criteria, such as reference to biodiversity, to indirect and cumulative ecological impacts, and to ecological monitoring (Thompson *et al*, 1997; Atkinson *et al*, 2000; Mandelik *et al*, 2005). Hence there is an inherent, recurring pattern of shortcomings in ecological impact assessments that go beyond differences in the procedural aspects of the scoping phase and merit further investigation.

In essence, ecological scoping should provide for information about the affected ecosystem and its components, and an interpretation of the proposal and its associated effects (Treweek, 1999). This is a formidable task taking into account the complexity of ecosystems and lack of basic ecological knowledge. Moreover, stringent time and budgetary constraints further limit the practice of the best available knowledge in ecological scoping. This raises fundamental dilemmas regarding how to maneuver between the need to thoroughly address ecological effects and making realistic demands.

This paper explores the issues and dilemmas incorporated in the process of ecological scoping. The aim is to promote the use of ecological scoping as a mechanism for addressing biodiversity conservation issues in EIA. Four key issues that need to be addressed are identified and the fundamental dilemmas they evoke discussed: biodiversity assessments; appropriate spatial and temporal scales; and cumulative ecological effects.

Recommendations as to how to address these key issues are provided and their applicability to the EIA process discussed, taking into account limitations and possible ways to overcome these, from scientific, procedural, and economic perspectives. Work on the Israeli case study is used as a backdrop, but the same dilemmas face all analysts who try to establish appropriate ecological scoping processes. Finally, the paper proposes a generic blueprint for ecological scoping.

The Israeli case study

Israel is a global hot spot of biodiversity (Yom-Tov and Tchernov, 1988) where fauna and flora are threatened by exceptionally rapid development, reflecting high population growth rate and density (approximately 600 people per km^2 in its northern half) and a continuous increase in standards of living (Central Bureau of Statistics, 1997). Progressive habitat loss and fragmentation, as a result of urbanization, suburbanization, and infrastructure development, are threatening most of Israel's ecosystems and driving many species to extinction (Perevolotsky

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and Dolev, 2002). Therefore, planning tools are critical for protecting Israel's biodiversity.

One of the most important tools in land-use planning in Israel is EIA (Brachya, 1993). Since EIA regulations came into effect in 1982, the number of EISs prepared have increased sharply from an annual average of seven in the 1980s, to 32 in the 1990s. This trend indicates a growing reliance of planning authorities on EISs as a tool for incorporating environmental considerations into plans.

The Israeli EIA system is based on a mandatory scoping process performed by the Ministry of the Environment (MOE) in consultation with relevant governmental and non-governmental organizations (depending on the type of plan and its location) (MOE, 1997). The scoping is case-specific and the guidelines, which take the form of a detailed checklist, refer to the environmental aspects, spatial scales, and specific analyses required. In most cases, these guidelines have to be approved by the planning committees to whom the EIA findings are ultimately referred.

In a recent study of the ecological component of the Israeli EIA system (Mandelik *et al*, 2002; 2005), a quantitative investigation of a representative sample of 52 EISs was undertaken in order to elucidate the factors that affect their scientific quality, that is, the objectivity, defensibility and application of the best available knowledge in ecological baseline assessments, impact evaluations and mitigation proposed. Using a uniform evaluation form consisting of 64 detailed criteria, various shortcomings were revealed in all main categories examined.

A simple scoring technique was applied to determine a quality score for the EISs and their guidelines. The guideline quality score was the most important factor in determining EIS quality scores, accounting for 81% of the variation ($r=0.9$) (Figure 1) (Mandelik *et al*, 2005). Hence, improvement of the EIA system hinges largely on more professional guidelines, reflecting thorough ecological understanding and practical experience in focusing the study to the most critical aspects that need to be addressed, while making scientifically and economically realistic demands. Similar conclusions have been reached for other EIA systems as well (Barker and Wood, 1999).

While ecological scoping clearly needs to be improved, doing so evokes several fundamental dilemmas, not necessarily encountered in other aspects of EIA. Four prominent issues in the performance of ecological scoping are identified and discussed from scientific, procedural, and economic perspectives, together with a presentation of a set of attributes that can help to incorporate these issues better into EIA. Table 2 highlights these key issues that are in need of better attention in ecological scoping, proposes how they could be assimilated in the EIA study, and gives the expected costs incurred in their implementation. Finally, a generic blueprint for ecological scoping is provided (Appendix 1).

Biodiversity assessments for EIA

Inventory and monitoring of all components of biodiversity (composition, structure and function at the

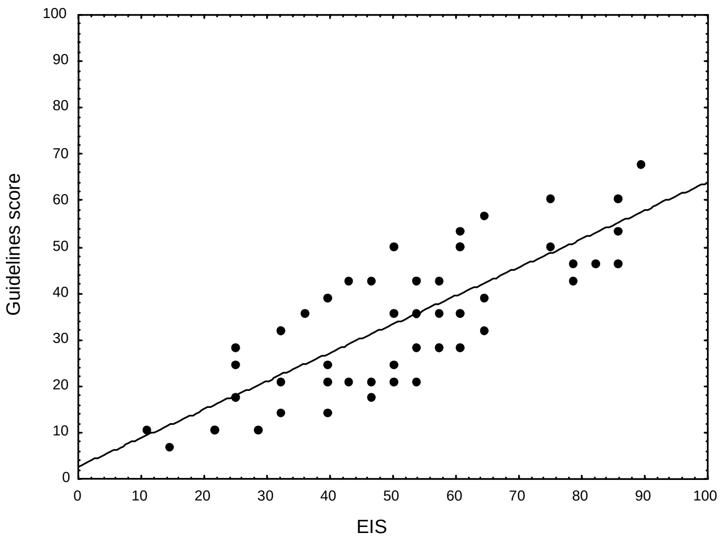

Figure 1. Correlation between EISs and guidelines quality scores
Source: Adapted from Mandelik *et al* (2005)

Table 2. Key issues in need of better attention in ecological scoping, their proposed attributes and translation into specific guidelines, and estimated direct (monetary) and indirect (delay in planning) costs incurred by each

Key issues	Attributes	Proposed guidelines	Direct monetary costs	Indirect (time delays) costs
Biodiversity assessments	Genetic diversity	Effective population size, heterozygosity, gene flow	Substantial	Substantial
	Species diversity	Species richness, diversity, population structure, demographic processes	Substantial	Substantial
	Ecosystem diversity	Vegetation density and layering, key spatial and physical features, biomass and productivity, succession	Moderate (using GIS and remote sensing)	Minor
Spatial scale	Direct effects Indirect and cumulative effects	Case specific- based on project type and location	Variable	Variable
Temporal scale	Preparation period Implementation Post-implementation	Case specific- based on project type and location	Variable	Variable
Cumulative effects	Cumulative spatial effects	Fragmentation indices, patch size distribution, connectivity	Moderate (using GIS and remote sensing)	Minor
	Cumulative environmental deterioration	Cumulative pollution, increased human accessibility	Variable	Variable

Note: List is not exhaustive

genetic, species and ecosystem level, following Noss (1990)) is an unrealistic task in most instances, especially in the course of EIA. Reliable shortcuts, that is, biodiversity indicators, are therefore needed. The combination of the diagnostic value of different surrogates and their costs in terms of money and time, will determine those suitable for use. The objective of good ecological scoping is to provide the minimum necessary data for informed decision-making (Slootweg and Kolhoff, 2001).

The most commonly used shortcut for species diversity assessments in conservation practice is the indicator taxa approach (Noss, 1990; Pearson, 1994; Caro and O'Doherty, 1999). Nevertheless, EIAs very rarely refer to it. A desirable indicator should be taxonomically well known and stable, have well known natural history, be readily surveyed and manipulated, be indicative of other taxa in the ecosystem, and exhibit strong habitat specialization and sensitivity to environmental changes (Pearson, 1994).

While ecological scoping clearly needs to be improved, doing so evokes several fundamental dilemmas, not necessarily encountered in other aspects of EIA

Most vertebrates fail to meet most of these requirements, while invertebrates, especially insects, generally fulfill many (Hilty and Merenlender, 2000). An overview of the faunal taxa most commonly referred to in EIAs demonstrates an opposite trend; birds and large mammals are commonly referred to, while invertebrates are very rarely mentioned (Mandelik *et al*, 2005). The main reasons for taxa selection in EIA studies are availability of data, legal obligations (protected species) and public appeal, while indicative abilities do not seem to play a major role.

Vegetation (vascular plants) is the most common taxon referred to in EIA studies (Thompson *et al*, 1997; Byron *et al*, 2000; Mandelik *et al*, 2005) although its indicative abilities may be limited since it does not necessarily correlate with faunal richness or may exhibit low correlation coefficients (Crisp *et al*, 1998; Panzer and Schwartz, 1998; Jonsson and Jonsell, 1999). Vegetation is also used to classify habitats and ecosystems, and delineate field surveys; structural characteristics of the vegetation are used as measures of habitat heterogeneity and correlate with different faunistic taxa (Tews *et al*, 2004). Hence, the use of structural characteristics of the vegetation in EIA can be regarded as an application of the indicator approach, though not explicitly.

Although ecological sampling procedures are a complex and debated issue (for instance, Sutherland, 2002), scientific grounds exist on which appropriate survey design can be based. These include the indicator taxa, keystone species, and umbrella species concepts for appropriate taxon selection (Caro and

O'Doherty, 1999), the species–area relationship, habitat heterogeneity, and measurement of seasonal variations for appropriate spatial and temporal scale considerations (Rosenzweig, 1995).

Developments in rapid biodiversity assessment, using higher taxa and morphospecies identifications (Oliver and Beattie, 1996; Kerr *et al*, 2000), can reduce the time and expertise required for taxonomic identification and be beneficial for EIAs. In addition, existing sources, especially national databases and museum collections, can be an important supplement to biodiversity assessment (Ponder *et al*, 2001), but they cannot replace in-situ surveys, as they lack the resolution and up-dated information needed for EISs.

At the ecosystem level, spatial characteristics of the landscape, such as size, shape, and connectivity of patches, cover types and patterns, easily and inexpensively obtained using GIS techniques (where such a system is already in place), can be used as indicators of diversity (EC, 1999; Noss, 1999) (Table 2). However, it remains to be tested whether these parameters correlate with diversity patterns at the species level (MacNally *et al*, 2002).

At the genetic level, measures such as allelic (gene) diversity, heterozygosity, and polymorphism can be used to assess population-level diversity (Noss, 1990), but are expensive, highly time consuming, and almost never used in EIA (for instance, Mandelik *et al*, 2005). Genetic databases with EIA-level resolution are mostly absent. The large investments necessary for the integration of genetic diversity in EIAs evokes a fundamental question of how relevant and appropriate it is. Is it reasonable to assume that genetic studies can take place in the time course of an average EIA? Even if the answer is 'yes', are they likely to produce the planning input necessary to justify the investment? It is important to make a distinction between a conceptual, holistic 'wish list' for ecological scoping (Slootweg and Kolhoff, 2003) and what is actually practical. In the current state of knowledge and resources, the feasibility of genetic studies in EIA is highly questionable.

In summary, despite a growing body of research on biodiversity assessment there is still a long way to go in knowing what to measure in order to have the information needed for decision-making (Noss, 1999). Often surrogate measures of biodiversity will have to be used with no *a priori* research in the specific ecosystem or habitat type dealt with. However, a growing body of ecological research on both biodiversity indicators and ecosystem processes can provide the grounds for an educated qualitative judgment that will suffice for most EIAs.

Nonetheless, biodiversity assessments (particularly at the species and gene levels) are expensive and take time; biodiversity assessment is expected to constitute the bulk of monetary costs in ecological assessments (Table 2). The robustness of the indicators used in EIAs should be determined based on two core criteria: their ability to represent patterns of diversity; and their ability to distinguish the most sensitive components (species, ecosystems or groups of people) that are in need of special attention, that is, in-depth evaluations of current state and severity of expected impacts.

Spatial and temporal scales

The spatial and temporal scales that should be addressed in ecological scoping are not clear and straightforward to determine. The spatial scale should go beyond the boundaries of the project, and encompass all the areas potentially affected, including those with indirect and cumulative impacts (Treweek, 1999). For example, a new road will impact the fauna and flora up to few hundred meters from the road (Forman and Alexander, 1998), a scale suitable for detailed biodiversity assessments. However, cumulative and indirect ecological effects such as fragmentation, cumulative habitat loss, or cascading effects of species displacement, clearly impact ecosystems on a much wider scale.

Similarly, to address the effects of the road on ecosystem structure and function using a keystone species, it will be necessary to look at the ecosystem scale. If the road transects a stream, a different scale should be determined, based on upstream and downstream characteristics. Hence, in most cases it would be erroneous to focus on a single predetermined spatial scale, as distance and shape of the impact area depend on specific issues and the location. This greatly confounds and complicates ecological scoping and the costs incurred by these evaluations will vary accordingly (Table 2).

There is an inevitable trade-off between the resolution of assessments and the spatial scale addressed. High-resolution assessments should be balanced against the area that can be surveyed. Trying to promote both simultaneously risks unacceptably high costs that may jeopardize the value of ecological scoping altogether. The appropriate balance between these two conflicting demands will have to be determined on a case-specific basis, according to the conservation value of the project site and its vicinity, relative to the wider surroundings, as determined by GIS analysis and aerial photographs (Haklay *et al*, 1998), where they are available.

Similar dilemmas arise when determining the appropriate temporal scales to be addressed. The timing and duration of ecological assessments have pivotal effects on the final evaluations. The timeframe of ecological baseline and impact assessments should take account of the full duration of the project, from preparation and implementation to decommissioning (if relevant) and follow-up (Treweek, 1999) (Table 2). Baseline assessments should reflect seasonality and multi-annual variation in ecosystem components; impact prediction should reflect long-term ecosystem processes, including delayed and long-term effects.

These are ideal standards, but are not necessarily optimal, as they do not take account of the substantial costs incurred by developers from the delays in the assessment process. It is not clear whether it is always justifiable to make such expensive and time-consuming assessments in an EIA, or whether there should be some differentiation between projects based on their expected impacts and the sensitivity of the potentially affected areas.

There is no easy way round this dilemma, but one possible solution would be to conduct a two-phase ecological scoping, where the first stage is used to screen projects that are expected to cause substantial ecological effects or affect highly sensitive areas. Only those singled out in the first stage would be required to continue and to conduct elaborate and more detailed baseline assessments and impact predictions. This idea is explored in the final discussion.

Cumulative ecological effects

Reference to cumulative effects is fundamental in realizing the full range of the project's potential consequences, since various stressors, while individually of little ecological significance, may have serious implications when considered collectively (Treweek, 1999). In addition, ecosystems may exhibit non-linear, interactive, or threshold responses to accumulated perturbations (Cocklin *et al*, 1992), imperiling the long-term viability of heavily developed ecosystems. Cumulative habitat loss and fragmentation are the major threats today to biodiversity (Terborgh, 1999).

Addressing cumulative effects requires appropriate spatial and temporal scales; the dilemmas raised in the previous section are relevant to this assessment as well. In addition, a major challenge is evaluating the ecological significance of cumulative effects, usually expressed at the ecosystem level (Thérivel *et al*, 1992). The sensitivity of ecological receptors (for instance, species, habitats) is a key factor to consider at this stage. Cumulative impacts are usually non-linearly magnified and difficult to model (Treweek, 1999). High levels of stochasticity and uncertainty hamper their evaluation.

A possible way to reduce the indirect cost of ecological scoping may be to promote a two-phase scoping process: a preliminary phase to differentiate between projects expected to have severe ecological effects and more moderate projects; and a more detailed second-stage scoping

Cumulative spatial effects, that is, cumulative habitat loss and fragmentation, are an exception and can be readily incorporated into the EIA, through parameters such as fragmentation indices, patch shape indices and patch frequency distribution (Noss, 1999), all readily achievable in the GIS era. In addition, by addressing existing stressors in the surrounding area, the potential for cumulative effects can be envisaged, though not evaluated or quantified.

Discussion and conclusions

The ultimate test of scoping is whether it is accepted by the planning committees that have to balance environmental and development goals. If scoping over-prescribes the ecological analysis, it might increase expenses and delays to a level where it will be rejected by the planning authorities. On the other hand, scoping that overlooks or disregards major ecological effects, and limits the spatial scope of assessment to the immediate vicinity of the plan, will have little scientific and conservation value.

Hence, there is an inherent tension incorporated in ecological scoping, that has to be balanced on a case-specific basis. While substantial direct and indirect costs might be inevitable if good ecological assessments are to be made, especially for appropriate biodiversity assessments (Table 2), there are ways to reduce these costs to some extent.

A possible way to reduce the indirect cost of ecological scoping may be to promote a two-phase scoping process. A preliminary phase would be used to differentiate between projects expected to have severe ecological effects and more moderate projects. This phase could be based on criteria commonly used for screening, for instance, type and magnitude of projects and sensitivity of the proposed location (Glasson *et al*, 1999), and could rely mostly on expert judgment for final ruling, in order to keep time delays and monetary costs minimal. Based on this, either a standard or a more detailed second-stage scoping would be conducted, in which the spatial and temporal scales and the specific issues that need to be addressed would be determined.

The main advantage of this approach is that it would create a differentiation between projects based on their expected ecological impacts, and would redirect the entire EIA process, and its associated costs, accordingly. By doing so it would reduce the inherent tension incorporated in ecological scoping between ecological and development goals. This approach would allow flexibility on a case-specific basis, as opposed to sector or location basis, which usually lack the resolution required for addressing ecological aspects.

At the preliminary stage, broad spatial scales should be preferred over high-resolution data. Hence, it should be based on ecosystem-level evaluations and cumulative (spatial) impact assessment, relying mostly on less costly and time-consuming GIS and remote sensing (if the required infrastructure is already in place). Detailed field surveys at the species or genetic level should be postponed to the second scoping phase, when their resolution and scales would be determined based on the ruling made in the first stage. Consequently, different costs would be inflicted on projects differing in their potential ecological effects, thereby enhancing the legitimacy of demands for comprehensive EISs when they are truly required. Nevertheless, this approach is theoretical and it remains to be tested in the actual planning arena.

The proposed procedural modifications still hinge on the improvement of the ecological aspects of the scoping process. Therefore a blueprint for ecological scoping is suggested (Appendix 1), based on previous research identifying the lacunae in ecological assessment in Israel (Mandelik *et al*, 2002; 2005). The proposed blueprint is relevant also to other countries, as Israel's EIA system is fairly typical (Feitelson, 1996); the major impediment for thorough ecological assessments in Israel, as in many other countries, is lack of basic ecological knowledge.

The blueprint highlights the importance of a broad, ecosystem-level perspective, using broad spatial scales, biodiversity (following Noss, 1990), ecosystem structure and function, and ecological interactions as proxies. The blueprint does not refer to societal aspects, being primarily focused on drawing scientifically good-practice guidelines from a conservation perspective. Societal issues, including equitable sharing, need to be addressed subsequently by the planning committees. Special attention was given to sensitive (that is, rare, threatened) species and habitats in both baseline information and impact evaluations.

A practical approach was applied, and, following discussions with EIA practitioners and ministerial officers, certain requirements were omitted from an earlier draft, such as reference to genetic diversity, as addressing these in the current state of resources is largely impractical. In addition, broad and vague requirements were avoided, such as reference to cumulative effects or to provision of goods and services, as these hold little practical value; instead these themes were decomposed to specific concrete requirements, for instance, cumulative habitat loss and fragmentation, changes in succession and dispersion processes. Aspects that should clearly be determined on a case-specific basis, such as spatial and temporal scales, were not referred to in the blueprint.

Although the blueprint is meant to be broadly applicable, it cannot replace the expertise and best judgment of an ecological advisor. On a case-specific basis and available knowledge an ecological advisor may find certain aspects redundant, and others needing to be augmented, and should act accordingly. The blueprint, however, can still serve in such cases as a basis for critical evaluation and discussion of the advisor's recommendations and judgments.

Appendix 1. A proposed blueprint for ecological scoping

1. General requirements

Consistency — baseline information, impact assessment and evaluation, and mitigation measures should be consistent in both spatial scales and issues addressed.

- Uncertainties and limitation — uncertainties and data deficiencies should be clearly stated.
- Presentation:
  - Use GIS to demonstrate pre- and post-implementation state
  - Present major ecological findings in the executive summary
  - List background and reference material.

2. Baseline information

Habitats

- Identification and distribution — describe the different habitats in the area, including their size, dominant plant communities, succession stage (if relevant), spatial distribution, and quality (that is, as examples of their habitat type).
- Rarity — indicate rarity of habitats and plant communities in a local to national and international scale.
- Wildlife resources — indicate the existence of water resources, wildlife breeding, feeding or resting sites in the area.

Fauna and flora

- Scope — based on a field survey, databases, and the literature describe the fauna and flora present in the area, including annuals, perennials, invertebrates, and vertebrates. Include quantitative estimates.
- Ecological requirements — indicate home ranges, habitat preferences, foraging and breeding requirements, sensitivity to anthropogenic disturbance of representative species or taxa (that is, indicator species, umbrella species, keystone species or other subset of species representing different habitats and guilds, or species similar in their exploitation of environmental resources).
- Conservation status — indicate endangered, endemic, red listed, protected, and peripheral species, and the degree of rarity and endangerment at a local to global scale.

Surrounding area

- Indicate proximity of the project to nature reserves and forests, ecological corridors, or other open landscapes and protected areas.
- Ecological sensitivity evaluation — refer to the ecological sensitivity of the region based on prior planning work (if present) and on the habitats, fauna and flora, reserves, and ecological corridors.

Field survey

- Scope — field survey of habitats, fauna and flora should be performed.
- Location — the survey should reflect the spatial heterogeneity of the area (among and within habitats).
- Timing — the survey should reflect seasonality.
- Transparency — timing, duration, methodology, and identity of the person who performed the survey should be clearly stated.

(*continued*)

Appendix 1 (*continued*)

3. Impact assessment and evaluation

Impacts during the construction

- Specify all potential ecological effects expected during construction.

Habitats

- Habitat loss — indicate the extent and type of habitat loss.
- Rare habitats — indicate the extent of rare habitat loss relative to the remaining habitat in the surroundings.
- Fragmentation — indicate the size, spatial configuration and continuity of the remaining habitats, and whether the plan could act as a barrier for dispersion.
- Conservation status — evaluate the significance of the effects mentioned previously in terms of rarity and endangerment level of habitats, on a local to global scale.
- Indicate potential impacts on key ecosystem processes (succession, dispersion, and so on).
- Wildlife resources — indicate potential impacts on wildlife water resources, feeding, breeding and resting sites.
- Indicate possible impacts on the surroundings — pollution, enhanced illumination, noise, facilitated access to people, and so on.

Fauna and flora

- Direct effects — indicate species that will be directly affected (inhabiting the area to be taken, based on data presented in the baseline).
- Invasive species — indicate the possibility of introducing alien invasive species and their possible effects on the native fauna and flora.
- Species of special conservation status (endangered, endemic, red listed, protected and peripheral) — indicate the number of individuals that will be directly affected, expected effects of habitat loss (relative to home range requirements, breeding or resting sites, food and water resources), and fragmentation (limited dispersion, increased mortality).
- Conservation status — evaluate the significance of the effects mentioned in the previous section in terms of rarity and endangerment level, on a local to global scale.
- Reversibility — indicate whether the impacts presented above are reversible or irreversible.
- Delayed effects — indicate any potential delayed and long-term impacts.

4. Mitigation and monitoring

- Indicate mitigation measures to address the effects described above, including the reasons for its selection, and evaluation of its expected efficacy.
- Indicate what impacts will be un-mitigated, that is, residual effects.
- Describe a long-term monitoring program to examine the efficacy of mitigation measures used, changes in richness and composition of the fauna and flora.

5. Alternatives

- The above-mentioned requirements should be presented for each of the alternatives suggested for the siting, planning, and operation of the plan.

References

Atkinson, S F, S Bhatia, F A Schoolmaster and W T Waller (2000), “Treatment of biodiversity impacts in a sample of US environmental impact statements”, *Impact Assessment and Project Appraisal*, 18(4), pages 271–282.

Barker, A, and C Wood (1999), “An evaluation of EIA systems performance in eight EU countries”, *Environmental Impact Assessment Review*, 19, pages 387–404.

Brachya, V (1993), “Environmental assessment in land use planning in Israel”, *Landscape and Urban Planning*, 23, pages 167–181.

Brachya, V, and U Marinov (1995), “Environmental impact statements in Israel and other countries: a comparative analysis” (in Hebrew), *Horizons in Geography*, 42–43, pages 71–78.

Byron, H J, J R Treweek, W R Sheate and S Thompson (2000), “Road developments in the UK: an analysis of ecological assessment in environmental impact statements produced between 1993 and 1997”, *Journal of Environmental Planning and Management*, 43(1), pages 71–97.

Caro, T M, and G O’Doherty (1999), “On the use of surrogate species in conservation biology”, *Conservation Biology*, 13(4), pages 805–814.

Central Bureau of Statistics, Israel (1997), “Projections of Israel’s population until 2020”.

Cocklin, C, S Parker and J Hay (1992), “Notes on cumulative environmental change, I: concepts and issues”, *Journal of Environmental Management*, 35, pages 31–49.

Crisp, P N, K J M Dickinson and G W Gibbs (1998), “Does native invertebrate diversity reflect native plant diversity? A case study from New Zealand and implications for conservation”, *Biological Conservation*, 83(2), pages 209–220.

EC, European Commission (1999), “Towards environmental pressure indicators for the EU: indicator definition” (EC, Brussels).

Feitelson, E (1996), “Some spatial aspects of environmental impact statements in Israel”, *Geoforum*, 27, pages 527–537.

Forman, R T T, and L E Alexander (1998), “Roads and their major ecological effects”, *Annual Review of Ecology and Systematics*, 29, pages 207–231.

Glasson, J, R Therivel and A Chadwick (1999), *Introduction to Environmental Impact Assessment* (UCL Press, London, 2nd edition).

Gray, I M, and G Edwards-Jones (1999), “A review of the quality of environmental impact assessments in the Scottish forest sector”, *Forestry*, 72(1), pages 1–10.

Haklay, M, E Feitelson and Y Doytsher (1998), “The potential of a GIS-based scoping system: an Israeli proposal and case study”, *Environmental Impact Assessment Review*, 18, pages 439–459.

Hilty, J, and A Merenlender (2000), “Faunal indicator taxa selection for monitoring ecosystem health”, *Biological Conservation*, 92, pages 185–197.

Jonsson, B G, and M Jonsell (1999), “Exploring potential biodiversity indicators in boreal forests”, *Biodiversity and Conservation*, 8, pages 1417–1433.

Kennedy, A J, and W A Ross (1992), “An approach to integrate impact scoping with environmental impact assessments”, *Environmental Management*, 16(4), pages 475–484.

Kerr, J T, A Sugar and L Packer (2000), “Indicator taxa, rapid biodiversity assessment, and nestedness in an endangered ecosystem”, *Conservation Biology*, 14(6), pages 1726–1734.

MacNally, R, A F Bennett, G W Brown, L F Lumsden, A Yen, S Hinkley, P Lillywhite and D Ward (2002), “How well do ecosystem-based planning units represent different components of biodiversity?”, *Ecological Applications*, 12(3), pages 900–912.

Mandelik, Y, T Dayan and E Feitelson (2002), “Ecological impact assessment in Israel: a review of environmental statements”, *Proceedings of the 21st Annual IAIA Conference, June 2001* (The Hague, The Netherlands).

Mandelik, Y, T Dayan and E Feitelson (2005), “Planning for biodiversity: the role of ecological impact assessment”, *Conservation Biology*, in press.

MOE, Ministry of the Environment (1997), “Environmental impact statements” (in Hebrew) (MOE, Jerusalem).

Morgan, R K (1998), *Environmental Impact Assessment: a methodological perspective* (Kluwer Academic Publishers, Dordrecht).

Noss, R F (1990), “Indicators for monitoring biodiversity: a hierarchical approach”, *Conservation Biology*, 4, pages 355–364.

Noss, R F (1999), “Assessing and monitoring forest biodiversity: a suggested framework and indicators”, *Forest Ecology and Management*, 115, pages 135–146.

Oliver, I, and A J Beattie (1996), “Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity”, *Ecological Applications*, 6(2), pages 594–607.

Panzer, R, and M W Schwartz (1998), “Effectiveness of a vegetation-based approach to insect conservation”, *Conservation Biology*, 12(3), pages 693–702.

Pearson, D L (1994), “Selecting indicator taxa for the quantitative assessment of biodiversity”, *Philosophical Transactions of the Royal Society of London B*, 345, pages 75–79.

Perevolotsky, A, and A Dolev (2002), *Endangered Species in Israel. Red list of threatened animals: vertebrates* (in Hebrew) (The Nature and Parks Authority and the Society for the Preservation of Nature, Jerusalem).

Pimm, S L, and P Raven (2000), “Extinction by numbers”, *Nature*, 403, pages 843–844.

Ponder, W F, G A Carter, P Flemons and R R Chapman (2001), “Evaluation of museum collection data for use in biodiversity assessment”, *Conservation Biology*, 15(3), pages 648–657.

Rosenzweig, M L (1995), *Species Diversity in Space and Time* (Cambridge University Press, Cambridge).

Slootweg, R, and A Kolhoff (2001), *Proposed Conceptual and Procedural Framework for the Integration of Biological Diversity Considerations with National Systems for Impact Assessment* (International Association for Impact Assessment, Netherlands).

Slootweg, R, and A Kolhoff (2003), “A generic approach to integrate biodiversity considerations in screening and scoping for EIA”, *Environmental Impact Assessment Review*, 23, pages 657–681.

Sutherland, W J (2000), *The Conservation Handbook: Research, Management and Policy* (Blackwell Science, Oxford).

Sutherland, W J (2002), *Ecological Census Techniques: a handbook* (Cambridge University Press, Cambridge).

Tanaka, A (2001), “Changing ecological assessment and mitigation in Japan”, *Built Environment*, 27(1), pages 35–41.

Terborgh, J (1999), *Requiem for Nature* (Island, Washington DC).

Tews, J, U Brose, V Grimm, K Tielborger, M C Wichmann, M Schwager and F Jeltsch (2004), “Animal species diversity driven by habitat heterogeneity/ diversity: the importance of keystone structures”, *Journal of Biogeography*, 31, pages 79–92.

Thérivel, R, E Wilson, S Thompson, D Heaney and D Pritchard (1992), *Strategic Environmental Assessment* (Earthscan, London).

Thompson, S, J R Treweek and D J Thurling (1997), “The ecological component of environmental impact assessment: a critical review of British environmental statements”, *Journal of Environmental Planning and Management*, 40(2), pages 157–171.

Treweek, J R (1996), “Ecology and environmental impact assessment”, *Journal of Applied Ecology*, 33, pages 191–199.

Treweek, J R (1999), *Ecological Impact Assessment* (Blackwell Science, Oxford).

Treweek, J R, S Thompson, N Veitch and C Japp (1993), “Ecological assessment of proposed road developments: a review of environmental statements”, *Journal of Environmental Planning and Management*, 36(3), pages 295–307.

UNEP/CBD/SBSTTA, UN Environmental Program/Convention on Biological Diversity/Subsidiary Body on Scientific, Technical and Technological Advice (2003), “Proposals for further development and refinement of the guidelines for incorporating biodiversity-related issues into the environmental impact assessment legislation or procedures and in strategic impact assessment: report on ongoing work”, UNEP/CBD/SBSTTA/ 9/INF/18.

Warnken, J, and R Buckley (1998), “Scientific quality of tourism environmental impact assessment”, *Journal of Applied Ecology*, 35, pages 1–8.

Yom-Tov, Y, and E Tchernov (1988), “Zoogeography of Israel”, in E Yom-Tov and E Tchernov (editors), *The Zoogeography of Israel* (Junk Publishers, Dordrecht).