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Biodiversity and ecosystem services in impact assessment – from components to services

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UNIVERSITY OF HELSINKI
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ACADEMIC DISSERTATION

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ABSTRACT

Ecological impact assessment focuses both on spatially bound biophysical environment and biodiversity as composition, structure, and key processes and on benefits of biodiversity gained through ecosystem services. It deals with allocation of space in complex situations characterised by uncertainty and conflicting values of actors. In the process of ecological impact assessment that forms part of environmental impact assessment (EIA) and strategic environmental assessment (SEA), the whole proposal of a project, plan, or programme; its targets; alternative options and their acceptability from a biodiversity standpoint; and knowledge of the biodiversity and ecosystem services it provides are shaped.

The analyses in this thesis examine the current practices of Finnish ecological impact assessment with respect to its substantive and procedural features and the roles of actors. The analyses utilise qualitative and semi-quantitative data from EIA and Natura 2000 appropriate assessment reports, statements of environmental authorities, other data produced via assessment processes, and actors' views related to ecological impact assessment. After analysis of the present shortcomings, constraints, and development needs, a tool taking into account fully current understanding and ecosystem services is developed to improve prevailing impact assessment practices.

The results of the analyses demonstrate that the knowledge base for the comprehensive ecological impact assessment in EIA, Natura 2000 appropriate assessment, and municipal land-use planning SEA is far from adequate. Impact assessments fail to identify the biodiversity at stake, what is affected, and how, and, as a consequence, the selection of biodiversity elements for assessment is unsystematic, superficial, or focused on the most obvious strictly protected species. The connection between baseline studies and impact prediction is loose; consequently, the predictive value of baseline studies is low, preventing effective mitigation and monitoring. There is also a tendency toward unnecessary detail at the expense of a broader treatment of biodiversity that would address ecosystem processes, interactions, and trends. Substantive treatment of biodiversity is often restricted to compositional diversity and at the species and habitat type level. Finnish ecological impact assessment does not take into account the value-laden nature of impact assessment. It is baseline-oriented and often seen as external and parallel to the actual planning and decision-making. Scoping practices reflect this separateness by outsourcing important value-bound significance determinations to individual ecology consultants instead of considering them an integral part of the planning process. Cumulative effects are hardly ever considered in Finnish ecological impact assessment practices.

The use of more sophisticated methods and tools than expert judgements and matrices is almost non-existent in Finnish ecological impact assessment practices, because of the planning environment lacking the time, resources, and skills for it. In addition, often a highly detailed treatment of biodiversity elements with complex tools is not necessary for achieving a holistic picture of the targets and impacts of an initiative. Therefore, an objective set for improvement in the knowledge grounding of ecological impact assessment has been the development of a relatively simple tool utilising already available data. Ecosystem services criteria and indicators were developed for target-setting, impact prediction, and monitoring, and these were tested in three processes of local master planning and regional planning. Timing constraints of data delivery; obstacles in data availability, quality, and consistency; and relative closeness of planning processes hampered the use

of indicators, but the tool nonetheless was experienced as beneficial by the testing teams overall. The future challenges facing use of the tool involve its independent utilisation by planners without support from researchers on different planning scales, collaboration and commitment of actors in setting targets for ecosystem services, and versatile use of data.

The other challenges in improvement of today's ecological impact assessment practices comprise finding a balance between broad-brush and detailed information individually for each planning situation; utilising, sharing, and mediating both knowledge within ecosystem-service-generating units and users' and beneficiaries' views of valued and prioritised ecosystem services; shifting from parallel linkage of impact assessment and planning towards planning- and decision-making-centred environmental assessment; supplying the necessary substantive and procedural requirements for ecological impact assessment in the EIA, nature conservation, and land-use and building legislation; placing stronger emphasis on scoping by strengthening the guiding role of authorities and reserving more time and resources for scoping by proponents and planners; generating specific cumulative impact assessment in EIA and SEA and improving that employed in Natura 2000 appropriate assessment by creating an iterative link; and fostering work-sharing between project- and plan/programme-level actors in identification of cumulative impacts.

Keywords: ecological impact assessment, biodiversity, ecosystem services, environmental assessment, EIA, SEA, spatial planning, Natura 2000 appropriate assessment

TIIVISTELMÄ

Luontovaikutusten arviointi paneutuu sekä paikkasidonnaiseen biofyysiseen ympäristöön ja luonnon monimuotoisuuden koostumukseen, rakenteeseen ja prosesseihin että ekosysteemipalvelujen kautta määrittäviin biodiversiteetin hyötyihin. Se käsittelee maankäytön ratkaisuja monimutkaisissa tilanteissa, joita luonnehtivat epävarmuus ja toimijoiden ristiriitaiset arvot. Ympäristövaikutusten arviointimenettelyn (YVA) ja suunnitelmien ja ohjelmien vaikutusten arvioinnissa (SOVA) luontovaikutusten arviointiprosessissa muokkautuvat koko hankkeen, suunnitelman tai ohjelman tavoitteet, vaihtoehdot ja niiden hyväksyttävyyden luonnon monimuotoisuuden ylläpidon näkökulmasta. Niin ikään tieto luonnon monimuotoisuudesta ja sen tarjoamista ekosysteemipalveluista määrittyy.

Tämä väitöskirja tarkastelee suomalaisen luontovaikutusten arvioinnin sisältöä, prosessia ja toimijoiden rooleja. Tutkimuksen laadullisten ja määrällisten analyysien aineistona on käytetty YVA- ja Natura-arviointiraportteja, viranomaisten lausuntoja arvioinneista, muuta arviointiaineistoa ja luontovaikutusten arvioinnin toimijoiden haastatteluaineistoja. Arviointikäytännön nykyisten puutteiden, rajoitteiden ja kehittämistarpeiden käsittelyn jälkeen kehitettiin ajantasaiseen tieteelliseen ekosysteemipalvelutietoon perustuva ekosysteemipalvelukriteeristö ja -indikaattorit.

Tutkimuksen mukaan luontovaikutusten arvioinnin tietopohja YVAssa, Natura-arvioinnissa ja yleiskaavoituksen luontovaikutusten arvioinnissa on kaukana kattavasta. Arvioinnit eivät onnistu tunnistamaan vaikutusten arvioinnin kannalta oleellisia biodiversiteetin piirteitä eli sitä, mihin ja miten vaikutukset kohdistuvat. Tämän vuoksi tarkasteltavien kohteiden tai ilmiöiden valinta on epäsystemaattista ja pintapuolista tai tarkastavaksi valikoituvat ilmeisimmät, tiukasti suojellut lajit. Luontoselvitysten ja vaikutusarvioinnin yhteys on heikko. Siten myös luontoselvitysten ennustearvo on heikko, mikä puolestaan estää tehokkaasti vaikutusten lieventämisen ja seurannan. Arviointikäytännössä on myös pyrkimys tarpeettomaan yksityiskohtaisuuteen holistisen, ekosysteemiprosesseihin, vuorovaikutussuhteisiin ja kehityssuuntia tunnistavan luonnon monimuotoisuuteen keskittyvän tarkastelun kustannuksella. Sisällöllisesti arviointi on usein keskittynyt vain biodiversiteetin koostumukseen kuvaten pääosin lajistoa ja luontotyyppejä. Suomalainen luontovaikutusten arviointi ei huomioi vaikutusten arvioinnin arvosidonnaista luonnetta ja on hyvin perusselvitysorientoitunut. Lisäksi arviointi nähdään usein suunnitteluprosessille erillisenä ja rinnakkaisena toimintona. Tätä erillisyyttä kuvastavaa arvioinnin kohdentamiskäytäntöä, jossa arvosidonnaiset päätökset siitä, mitä ja miten selvitetään ja arvioidaan, ulkoistetaan yksittäisille luontokonsulteille sen sijaan, että arvioinnin kohdentamista käsiteltäisiin osana suunnitteluprosessia. Kasautuvien vaikutusten arviointi on hyvin niukkaa.

Asiantuntija-arvioita ja arviointimatriiseja monimutkaisempien menetelmien käyttö luontovaikutusten arvioinnissa on lähes olematonta, koska arvioinneissa ei ole aikaa, resursseja eikä asiantuntemusta tähän. Sitä paitsi hyvin yksityiskohtainen biodiversiteetin käsittely monimutkaisin menetelmin on usein tarpeetonta holistisen näkemyksen saavuttamiseksi hankkeen, suunnitelman tai ohjelman tavoitteista ja vaikutuksista. Tutkimuksen tavoitteena luontovaikutusten arvioinnin tietopohjan parantamiseksi olikin suhteellisen yksinkertaisen, olemassa olevien tietoja hyödyntävän työkalun kehittäminen. Ekosysteemipalvelukriteeristö- ja indikaattorit kehitettiin tavoitteenasettelua, vaikutusten arviointia ja seuranta varten ja niitä testattiin kolmessa yleiskaavoitusta ja maakuntasuunnittelua koskevassa suunnitteluprosessissa. Tiedon toimittamisen ajoitukseen, tiedon saatavuuteen, laatuun ja yhdenmukaisuuteen liittyvät ongelmat ja suunnitteluprosessien avoimuuden puute häiritsevät kriteeristön ja indikaattorien käyttöä, mutta kokonaisuudessaan työkalu koettiin hyödylliseksi testauksessa. Kriteeristön ja indikaattorien käytön tulevaisuuden haasteet koskevat monipuolista tiedon käyttöä, työkalun itsenäistä käyttöä eri suunnittelutasoilla ilman tutkijoiden tukea ja suunnittelun ja arvioinnin toimijoiden sitoutumista ekosysteemipalvelutavoitteiden määrittelyyn.

Muita luontovaikutusten arvioinnin kehittämisen haasteita ovat tasapainon löytäminen yleispiirteisen ja yksityiskohtaisen tiedon välillä kussakin yksittäisessä suunnittelu- ja arviointitilanteessa; ekosysteemipalveluita tuottavien biodiversiteetin piirteiden ja palveluita hyödyntävien tai niistä hyötyvien toimijoiden näkemysten ja priorisointien hyödyntäminen, jakaminen ja yhteensovittaminen; siirtyminen vaikutusarvioinnin ja suunnittelun erillisyydestä kohti suunnittelu- ja päätöksentekokeskeistä vaikutusten arviointia; luontovaikutusten arvioinnin sisällön ja esittämisen tarkempi määrittäminen YVA-, luonnonsuojelu- ja maankäyttö- ja rakennuslainsäädännössä; arvioinnin kohdentamisen painotus vahvistamalla ohjaavien viranomaisten roolia sekä lisäämällä hankkeista ja suunnitelmista ja ohjelmista vastaavien panostusta kohdentamiseen; kasautuvien vaikutusten arvioinnin edistäminen YVAssa ja maankäytön suunnittelun vaikutusten arvioinnissa ja sen parantaminen Natura-arvioinneissa luomalla iteratiivinen yhteys ja työnjako eri suunnittelutasojen toimijoiden välille kasautuvien luontovaikutusten arvioinnissa.

Asiasanat: luontovaikutusten arviointi, luonnon monimuotoisuus, ekosysteemipalvelut, ympäristövaikutusten arviointi, YVA, SOVA, maankäytön suunnittelu, Natura-arviointi.

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Helsinki, March 2012

Tarja Söderman

GLOSSARY

AA	Appropriate assessment, the part of Natura 2000 assessment including scoping and actual assessment
AP	Assessment programme of EIA (scoping report)
AR	Assessment report of EIA (corresponding environmental impact statement EIS)
Biodiversity aspects	The composition, structure, and key processes of biodiversity
Biodiversity elements	Specific elements of biodiversity aspects addressed in ecological impact assessment
CBD	Convention of Biological Diversity
Ecological impact assessment	Impact assessment addressing biodiversity and ecosystem services
Ecosystem services	Subjectively valued benefits to humans produced by biodiversity
EA	Environmental assessment including both environmental impact assessment and strategic environmental assessment
EIA	Environmental impact assessment
EIS	Environmental impact statement, report describing results of environmental impact assessment
Initiative	Project, plan, programme, or policy
MSSS	Monitoring system of spatial structure, data system including spatial information in 250 x 250 metre grid format from national databases
Natura 2000 assessment	Assessment process of impacts on Natura 2000 sites of the European protected areas network
SEA	Strategic environmental assessment
VEC	Valued ecosystem component, a biodiversity element chosen to be addressed in environmental assessment

LIST OF ORIGINAL ARTICLES

This dissertation consists of a summary and the following articles reprinted with the kind permission of the publishers:

Söderman, T. (2005). Treatment of biodiversity issues in Finnish environmental impact assessment. *Impact Assessment and Project Appraisal* 22:2, 87–99.

Söderman, T. (2006). Treatment of biodiversity issues in impact assessment of electricity power transmission lines: A Finnish case review. *Environmental Impact Assessment Review* 26:4, 319–338.

Söderman, T. (2009). Natura 2000 appropriate assessment: Shortcomings and improvements in Finnish practice. *Environmental Impact Assessment Review* 29:2, 79–86.

Söderman, T. & Saarela, S.-R. (2010). Biodiversity in strategic environmental assessment (SEA) of municipal spatial plans in Finland. *Impact Assessment and Project Appraisal* 28:2, 117–133.

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Article I is a sole contribution by Söderman.

Article II is a sole contribution by Söderman.

Article III is a sole contribution by Söderman.

Article IV was developed jointly by the two authors, with Söderman as lead author. Söderman and Saarela contributed the research idea and analytical approaches as well as the interpretation of results on the basis of the analyses. Saarela collected the data. Söderman and Saarela contributed the writing process, led by Söderman.

Article V was developed by the four authors together, with the lead of Söderman, who contributed the research idea. Söderman, Kopperoinen, Yli-Pelkonen, and Shemeikka contributed the analytical approaches. Söderman contributed the interpretation of the results and the writing process. Kopperoinen and Shemeikka contributed the maps, and Yli-Pelkonen contributed boxes 1–3.

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1 Introduction

Biodiversity, the variety of ecosystems, species, and genes, is recognised as one of the critical elements for human existence, as it provides vital goods and services such as food, carbon sequestration, or wastewater purification. The rate of loss of biodiversity is considered to have passed its safe boundaries already. Notwithstanding large uncertainties linked with complexity of ecological systems and also lack of consensus on distinct cause-and-effect relationships and the true position of thresholds, it can be said with some confidence that Earth cannot sustain the current loss of biodiversity without reduction of its capacity to provide useful services (Rockström et al. 2009; TEEB 2010). In the EU, only 17% of the habitats and species assessed have a favourable conservation status – meaning that their natural range and the areas habitats cover ensure the habitats’ long-term maintenance and the species maintain themselves on a long-term basis – on account of major pressures and drivers causing biodiversity loss through habitat fragmentation, degradation, and destruction due to land-use change (European Environment Agency 2010). In Finland, the first assessment of threatened habitat types (Raunio et al. 2008) demonstrated that 51% of all habitat types are threatened; in Southern Finland, the proportion is as high as 66%. Similarly, the latest red-listing of species in Finland revealed that 10.5% of species are threatened (Rassi et al. 2010). Biodiversity loss occurs at the local and regional level but can have global effects – for example, in terms of capacity to adapt to climate change.

Increasing evidence of decreased biodiversity has put halting the loss of biodiversity high on the political agenda. The 10th meeting of the Conference of the Parties (COP) to the Convention on Biological Diversity, in Nagoya in October 2010, adopted a new 10-year global strategic plan to combat biodiversity loss in 2011–2020 (CBD 2010). In March 2011, the EU made two biodiversity commitments (European Commission 2010). The first is the headline target of halting loss of biodiversity and

degradation of ecosystem services by 2020. The second is the 2050 vision to protect, value, and restore biodiversity and the ecosystem services it provides, for their contribution to human well-being and economy and for avoidance of catastrophic changes caused by loss of biodiversity. The period for the previous EU biodiversity goal, set in 2001, ended in 2010, with the target of halting the biodiversity loss by 2010 not having been reached. Also, work did not tackle the three most significant drivers of change, two of which are strongly related to spatial planning: land-use change and over-exploitation / non-sustainable use of resources (European Commission 2011a). In addition, the EU’s Communication in January 2010 (European Commission 2010) saw as one of the serious shortcomings the neglect of provision of the ecosystem services outside protected areas. Furthermore, besides the above-mentioned previous strategic requirements (CEC 2006) of nature and environmental impact assessment directives (CEC 1979, 1992, 1997, 2001) applied in infrastructure development and spatial planning in consideration of alternatives and prevention and the reduction of negative effects on biodiversity, the Communication (European Commission 2010) emphasised the need for improvement in developing and investing in green infrastructure, for the interconnected green network besides the protected areas. In light of land-use-linked priorities in European biodiversity policy, a more coherent approach to development and spatial planning and to biodiversity impact assessment and knowledge production is called for. The European Union’s new biodiversity strategy was adopted in May 2011 (European Commission 2011b). While the previous, far-reaching strategy, from 2006, included 160 individual measures, the new strategy focuses only on six targets accompanied by 20 specific actions. It gives increased attention to ecosystem services, reflecting their importance to economy and human well-being, and emphasises that reaching the targets depends on the action and actors on multiple spatial scales: EU, national, regional, and local levels. Ecosystem services have received more and more attention since the publication of the Millen-

nium Ecosystem Assessment (MA 2003). The second target of the strategy has to do with better protection for ecosystems and more use of green infrastructure, a potential that at present is regarded as largely untapped in climate change adaptation (European Commission 2011b), and it is required that management authorities ensure that the impact on natural areas and land use is fully examined in their appraisal of all infrastructure projects (European Commission 2011c).

At the time of writing of this summary, the Finnish national biodiversity strategy ‘Saving Nature for People, National Strategy and Action Plan for Conservation and Use of Biodiversity in Finland 2006–2016’ (Heikkinen 2006) was renewed to meet the EU’s goal of halting biodiversity loss by 2020. This renewal was preceded by open public discussion on the Internet during March–April 2011 (Miten pysäytetään luonnon köyhtyminen? 2011). One of the central themes raised in the discussion was protected areas, especially with respect to how they do not aid in safeguarding biodiversity if nature outside them is used non-sustainably. Another central theme was the need for improvement in the legislation such that it addresses biodiversity issues in a more holistic way – not only with measures of the Nature Conservation Act (1996) but through development of measures covering land use and spatial planning (Land Use and Building Act 1999).

Environmental assessment has been around for more than 40 years, but biodiversity is still a relative newcomer on the global and European environmental scene (Rajvanshi et al. 2010). Thus far, regardless of political goals and public concerns, biodiversity has been poorly represented in environmental assessment and decision-making. It has been considered to be either too trivial or too difficult a subject to deal with, irrespective of the existing internationally acknowledged objectives and approaches for managing biodiversity (CBD 1992, 1999, 2002, 2004). Consequently, the problem is, moreover, to translate these objectives and approaches for environmental assessment to work in practice (Slootweg et al. 2010). Furthermore, environmental assessment, especially strategic

environmental assessment, has hitherto taken advantage of almost none of the opportunities provided by the concept of ecosystem services to translate biodiversity into human benefits recognisable in decision-making (van Beukering and Slootweg 2010).

This thesis has a European perspective – it explores environmental assessment from the European viewpoint, comparing the national practices mostly to those of other European countries and to EU legislation and guidelines in combination with international academic literature and best-practice approaches. In addition, the thesis approaches the subject of maintenance of biodiversity through the lens of the broad concept of biodiversity. It emphasises the basic purposes of environmental assessment: ensuring avoidance of negative effects and/or enhancing the positive influence of policies, programmes, plans, and projects on biodiversity. This is dealt with by examining the shortcomings and opportunities for improvement in present environmental assessment practices. At the same time, the work underscores the need for holistic approaches and knowledge production for decision-making in spatial planning, to ensure the sustainable use of spatially bound biodiversity and the ecosystem services it creates. It does so by developing planning and assessment methods, concentrating not on single species and habitats as components of biodiversity but on the whole array of services provided by biodiversity, and addressing the variety of actors on various spatial and temporal scales – here, there, and in the future.

2 The aim of the thesis

This thesis elucidates how biodiversity has been treated in impact assessment for 1) individual projects and 2) preparation of plans and programmes in the form of spatial planning. In this, I aim to find out whether and how Finnish ecological impact assessment practices reflect the international development of treatment of biodiversity and ecosystem services in environmental impact assessment (EIA) and strategic environmental assessment (SEA) and reflect also the conceptual shift of the general

theoretical embedding from natural-sciences-based ecosystem components' preservation toward more sustainability-oriented ecosystem services valuation. At a more practical level, I aim to identify, in detail, characteristics of ecological impact assessment practices in Finland and study how they comply with the requirements of national and EU-level legislation and with internationally reported best-practice views as set forth in the environmental assessment literature. In addition, I aim to identify shortcomings in Finnish ecological impact assessment practices and determine the underlying reasons for them. My purpose is to build on these findings to present broad recommendations and challenges in moving toward better ecological impact assessment and promotion of ecological sustainability. The findings and results applied to these ends are reported in detail in *articles I, II, III, IV, and V*. In this thesis, I draw them together by answering four main research questions:

1. What kinds of approaches have been used to generate a **knowledge basis** for impact assessment, and what kind of conceptual understanding of biodiversity and ecosystem services has developed on their basis?

2. How is the ecological **impact assessment process** structured in Finland in EIA, Natura 2000 assessment, and local master planning SEA, and do their present outcomes form a coherent and adequate basis for assessment of impacts on biodiversity?

3. What are the roles of **actors** in ecological impact assessment, what **forms** of communication and co-operation do they demonstrate, and what are their **views** on ecological impact assessment?

4. How can ecological **sustainability** be promoted in a Finnish setting **through impact assessment** that takes into account current understanding of biodiversity and ecosystem services and also involves improvement to the prevailing impact assessment practices?

3 Theoretical and conceptual framework

3.1 Background on environmental assessment of initiatives and their impact on biodiversity

Environmental assessment has been defined as a planning tool or a systematic process helping project-developers to improve their projects and authorities to improve their policies, plans, and programmes (Wathern 1988; Fischer 2007) by examining – in more detail, by identifying, estimating, and evaluating (Vanclay and Bronstein 1995) – the environmental consequences of proposed actions in advance and integrating environmental considerations into the planning process.

Environmental assessment has the character of a regulatory norm, with procedural norms dating back to 40 years ago with the enactment of the US National Environmental Policy Act (NEPA 1969, cited by Jay et al. 2007), which established the first legislative requirement for assessment of potential environmental impacts of development actions. Project-level impact assessment regulations are called environmental impact assessment, and assessment related to preparation of plans, programmes, and policies is termed strategic environmental assessment. In the 1970s, many developed nations introduced formal requirements for EIA (Sadler 1996). In 1985, the EIA Directive (CEC 1985) established minimum requirements for application of environmental impact assessment of projects in member states of the EU. By the early 1990s, more than 40 countries had legislated frameworks for EIA (Robinson 1992). The first international Earth Summit created an international law and policy structure promoting the use of environmental assessment (Rio Declaration on Environment and Development 1992, Principle 17). Now, in the early 2000s, more than 100 countries have implemented EIA as a legal and institutional force (Petts 1999; Wood 2003).

In the 1990s, it was noted that EIA was of greater assistance in reducing environmental

problems at source if used earlier in the decision-making process, and SEA was developed to identify the environmental consequences of higher-level planning (Therivel and Partidário 1996), with further integration of principles of EIA into the higher-level planning that set the background for EIA (Wood 2003). Methodologies and practice of more strategically oriented impact assessment for plans, programmes, and policies developed. Rather than to replace EIA, SEA was meant to complement it – but not just as an extension of EIA – and it has evolved its own distinct approaches and techniques (Fischer 2007). SEA took two lines of evolution from the 1990s: an appraisal-inspired, objective-led approach derived principally from policy appraisal processes (and political science) applied more at the policy level (e.g., to legislative proposals) and an EIA-inspired, baseline-led approach applied more at the plan and programme level (Partidário 1996; Devuyst 1999; Smith and Sheate 2001; Sheate et al. 2003; Pope et al. 2004; Fischer 2007). Partidário (2007) argues that the two main schools of SEA have recently become even more distinct. The two approaches can also be complementary, forming a combination wherein the objective-led approach is strengthened via improved baseline knowledge and public participation (Sheate 2001). The 2001 SEA Directive (CEC 2001) established the legal requirements for SEA in the EU.

The ‘environmental’ aspect of the assessment has been interpreted as having primarily nature-based considerations but also blending in environmental, social, and even economic aspects. In the early days of EIA, the term ‘environment’ was largely understood to refer to bio-physical systems that make up the ‘natural environment’. One of the stated original purposes was ‘to promote efforts which will prevent or eliminate damage to the biosphere’ (NEPA, Section 2, as cited by Jay et al. 2007). It has been pointed out that early EIA neglected consequences of proposals for human health and well-being, resulting in neglect of key social issues (Morgan 1988; Treweek 1999; Hildén 2000). Nevertheless, NEPA already included a goal of integration for both nature and man, as

well as an extended temporal perspective for sustainable development that would come to characterise later debates on environmental assessment (Wallington et al. 1997). NEPA aims to ‘create and maintain conditions under which man and nature can exist in productive harmony and fulfill the social, economic and other requirements of present and future generations of Americans’ (NEPA, Section 101(a), cited by Jay et al. 2007). Accordingly, neglect for societal matters has been suggested as having brought about emphasis on procedural characteristics and disregard for substantive matters in early assessment practice, rather than focus on the inherent character of environmental assessment *per se* (Sheate 2003; Jay et al. 2007). The 1990s understanding of environmental assessment highlighted both ecological and human aspects. For instance, Sadler (1996) defined EIA as ‘a process of identifying, predicting, evaluating and mitigating of bio-physical, social and other effects of proposed projects and physical activities prior to major decisions and commitments being made’.

Sadler and Verheem (1996) describe SEA as ‘a systematic process for evaluating the environmental consequences of [a] proposed policy, plan or programme in order to ensure they are fully included and appropriately assessed at the earlier appropriate stage of decision-making on par with economic and social considerations’. Brown and Therivel (2000) emphasise holistic understanding of the environmental and social implications of a policy proposal. The SEA Directive lists ‘environmental’ impacts as effects on biodiversity, population, human health, fauna, flora, soil, water, air, climatic factors, material assets, cultural heritage (including architectural and archaeological heritage), landscape, and interrelationships among these factors (CEC 2001, Annex I). The understanding of ‘environment’ applied in the SEA Directive includes impact on natural environment and social elements but not economic elements. However, there has been a long period in the literature that involved methodical development of equal treatment of three dimensions, with a ‘triple bottom line’, according to which approach integration is necessary and it would be

difficult to safeguard environmental resources without consideration of the social and economic perspective (Smith 1993; Eggenberger and Partidário 1999; Devuyt 1999; Brown and Therivel 2000; Partidário and Clark 2000; Eales et al. 2005; Dalal-Clayton and Sadler 2005).

By contrast, Morrison-Saunders and Fischer (2006), Kidd and Fischer (2007), Wallington et al. (2007), Bina (2007), and Jackson and Illsey (2007) hold an opposite opinion: that SEA's emphasis should rest on a pillar of environmental sustainability, instead of on integration of everything, and that SEA should have a constructive relationship with other appraisal processes, such as sustainability appraisal (SA) (e.g., George 2001), sustainability impact assessment (SIA) (e.g., Helming 2008), and integrated assessment (IA) (e.g., Lee 2006). Although Nilsson (2009) argues against this on the basis that it is necessary from a practical planning and policy-making viewpoint to treat all sustainability perspectives as integrated instead of applying a separate natural environmental focus in SEA, I agree with Wallington et al. (2007) that maintaining a separate natural environment focus – but only if that focus is kept rather broad, with the concept of biodiversity and ecosystem services incorporating both biophysical environment and human benefits and values attached to it – gives refreshing clarity to the substantive purpose of EIA and SEA against which also practical experiences can be weighed.

Slootweg et al. (2006) differentiate among perspectives in SEA along a spectrum ranging from assessments with substantive focus on the biophysical environment to those with a broad triple-line sustainability focus. With a strong biophysical environment focus, SEA tends to consider ecological elements from a nature conservation perspective, segregating conservation from economic and social development and concentrating on allocation of protected areas. This was a prevalent approach in treatment of ecological aspects of planning in the Nordic countries until the 1990s (Erikstad et al. 2008; Söderman 2006). According to this approach, ecological impacts are often regarded as just another category of impact, with ecological

impact assessment being used as a sub-discipline of environmental assessment alongside health impact assessment, economic impact assessment, social impact assessment, etc. Such partitioning by impact category can result in neglect of important links and interrelationships (Treweek 1999). With a strong focus on sustainability, biophysical environment is seen as providing for social and economic development and both social/economic and biophysical environments are seen as complementary to it. The third perspective is a merged approach of sector-based and integrated approaches including the biophysical environment as a provider of multiple, simultaneous benefits for humans across boundaries of geographical areas that are not clearly defined (Slootweg et al. 2006). Because ecological systems and functions are spatially bounded, along a continuum of land uses, some areas can be valued as more important for safeguarding human benefits than others might. Therefore, clarifying the substantive purpose of environmental assessment does not mean that the environment should have less weight than other concerns in SEA. On the contrary, it helps to avoid continued neglect for traditionally undervalued considerations of ecological systems and functions (Gibson 2006; Termorshuizen et al. 2007). It contributes to response to the reported failure of impact assessment frameworks in balancing sustainability, a failure said to stem from the dominance of socio-economic priorities in the prevailing planning traditions and cultures (Kørnøv and Thissen 2000; Hilding-Rydevik and Bjarnadóttir 2007; Nykvist and Nilsson 2009). Sadler (1999) argues that 'environmental impacts are at the core of sustainability concerns'. Sadler (1996) lists sharpening of environmental assessment as a tool for sustainability assurance as one important challenge in shifting the scale of assessment to focus on cumulative effects, interactive forms of public involvement, and incorporation of environmental assessment into decision-making at all levels.

Over the last two decades, the term 'biodiversity' has seen widespread use to describe ecological phenomena, especially in relation to preservation and management of natural en-

vironments. It is used as a broad political term meaning ‘the life on Earth’ and in a more scientific and technical sense (Wilson 1988; Noss 1990). Among the criticisms is that definitions of biodiversity are vague (e.g., Heywood and Baste 1995; Gaston 1996; Takacs 1996). Because of the multitude and vagueness of the definitions, it has been difficult to define and interpret biodiversity in environmental assessment practice (Slootweg 2005; Wegner et al. 2005; Wale and Yale 2010). The most commonplace definition of biodiversity in impact assessment is that found in the Convention on Biological Diversity (CBD 1992), which defines biological diversity as ‘the variability among living organisms from all sources, inter alia, terrestrial, marine and other aquatic ecosystems and ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems’. Byron (2000) sees the term ‘biodiversity’ as including the concept of sustainable use as a core component.

The Convention on Biological Diversity calls for the use of EIA and SEA procedures to ensure that the effects of development are assessed and taken into consideration (CBD 1992, 2002). The EIA Directive (CEC 1985) specifies that impacts on fauna and flora need to be considered. The SEA Directive (CEC 2001) specifies that biodiversity as well as flora and fauna must be part of the assessment. One form of assessment concentrates solely on biodiversity impacts. It requires impact assessment for projects, plans, or programmes affecting Natura 2000 sites designated on the basis of the directive on conservation of natural habitats and wild flora and fauna (CEC 1992), referred to below as the Habitats Directive. The Natura 2000 sites form an EU-wide network of protected sites that is based on the Habitats Directive and the Birds Directive (CEC 1979). The impact assessments are called appropriate assessments (Scott Wilson et al. 2006; Dodd et al. 2007; Kunzman et al. 2007; Therivel 2009), but on a national level, various terms are used in the individual EU member states (Peterson et al. 2010). These assessments are part of the EIA, SEA, and land-use planning process as sub-processes, or they may be individual parts of

permit consideration not connected to any other assessments (European Commission 2009a, 2009b). In most cases, however, they are a part of these broader assessment processes.

Slootweg et al. (2001), Slootweg and Kolhoff (2003), Slootweg et al. (2006), Slootweg (2010), and Slootweg and Molliga (2010) argue that operationalisation of biodiversity in environmental assessment will need to concentrate on the functions provided by biodiversity, the use and non-use values of the functions, and the impacts of biophysical and social changes on these functions and values. They present a general conceptual framework for impact assessment (see Figure 1), wherein physical, social, and to some extent economic interventions lead to biophysical and social changes that can result in higher-order changes. Some social changes can also lead to biophysical changes. Biophysical changes may influence several aspects of biodiversity, seen as

- i. composition (what is there) from the gene level, through species and ecosystems, to landscape level,
- ii. structure (how it is organised in space and time) (horizontal and vertical structure) and time (e.g., seasonal nature), and
- iii. key processes (physical, biological, biophysical, or human) that are important for its creation and maintenance (see also Noss 1990 for functional biodiversity).

Changes in these elements can have an impact on the ecosystem services provided through biodiversity. Slootweg et al. (2001) and Slootweg and Kolhoff (2003) call these ‘functions valued by society’, and Slootweg (2005) calls them ‘functions of biodiversity’. They have also been defined as ‘the benefits human populations derive, directly or indirectly, from ecosystem functions’ (Costanza et al. 1997) or as ‘those ecosystem functions that are currently perceived to support and protect human activities of affect human well-being’ (Barbier et al. 1994). The Millennium Ecosystem Assessment (MA 2003, 2005) defines these as ecosystem services which are ‘benefits that people obtain from ecosystems’ and emphasises how biodiversity is used and valued by society. It trans-

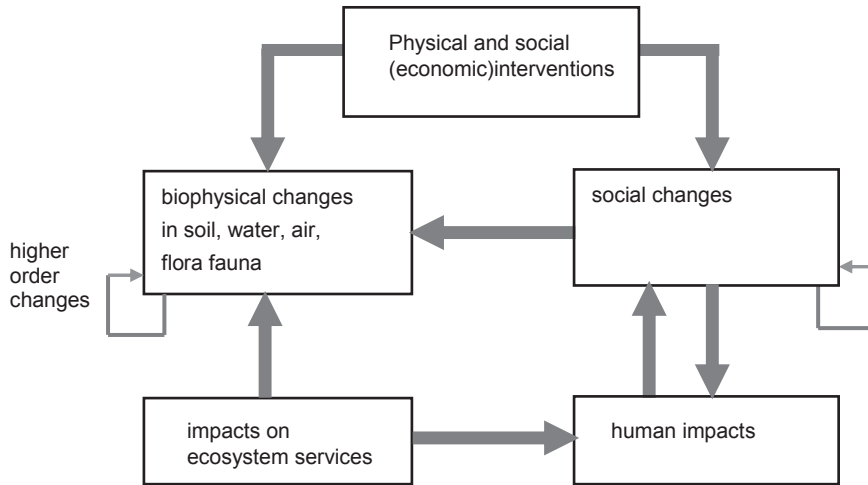


Figure 1. Conceptual framework for impact assessment concerning biodiversity (Based on Slootweg et al. 2006).

lates biodiversity into provisioning (e.g., food, water, fibre, and fuel), regulating (e.g., climate regulation, water, and disease-related), cultural (e.g., spiritual, aesthetic, recreation, and education), and supporting (e.g., primary production and soil formation) services to human well-being. Slootweg and Mollinga (2010) delineate one more service type, carrying services, which provide space, a substrate or a backdrop for human activities, with an example being water as a substrate for navigation. Costanza et al. (1997) identified 17 major categories of ecosystem services, and de Groot et al. (2002) identified 32 ecosystem services, including biological, physical, aesthetic, recreational, and cultural services. Examining literature from the 1990s and 2000s, Niemelä et al. (2010) identified 16 ecosystem services and their generating units (vegetation, micro-organisms, forests, etc.) in urban environments:

- i. provisioning services: 1) timber products; 2) food: game, berries, and mushrooms; and 3) soil and fresh water,
- ii. regulating services: 4) regulation of microclimate at the street and city level, 5) gas cycles: O₂ production and CO₂ consumption, 6) carbon sequestration and storage, 7) habitat provision, 8) purification from air pollution, 9) noise cushioning in built-up areas and by transportation chan-

nels, 10) rainwater absorption: balancing of storm-water peaks, 11) water filtration, 12) pollination: maintaining flower populations and food production, and 13) humus production and maintaining of nutrient content, and

- iii. cultural services: 14) recreation for urban dwellers; 15) psycho-physical and social health benefits; and 16) science education, research, and teaching.

According to the assessment framework (Slootweg et al. 2006; see also Figure 1), impact on ecosystem services will lead to a change in the valuation of these ecosystem services by various stakeholders in society, thus affecting human well-being. How and whether ecosystem services are valued by society/stakeholders is completely dependent on societal circumstances (Slootweg and Kolhoff 2003). People may respond to these changes in the value assigned to ecosystem services and act accordingly, bringing about new social changes in so doing. Thinking in line with the Millennium Ecosystem Assessment (MA 2003) approach parallels this, indicating that ecosystem services can be affected by drivers of change. These drivers might be natural or human-induced, direct and indirect. Direct drivers of change can be identified and measured and include the following

groups: changes in land use and land cover; fragmentation and isolation; extraction, harvesting, or removal of species; external inputs such as emissions, effluents, or chemicals; disturbance; introduction of invasive, alien, and/or genetically modified species; and restorations. Indirect drivers of change can include demographic, economic, socio-political, cultural, and technological processes or interventions. They are diffuse societal processes that influence or even govern direct drivers of change. Therefore, identifying chains of cause and effect is essential in environmental assessment.

Slootweg et al. (2006) highlight that the concept of ecosystem services is a strong tool for impact assessment, as it provides a means to translate biodiversity into aspects of human well-being. It links prerequisites of and threats to these services into the assessment framework (see *Article IV*, Figure 1). The concept of ecosystem goods and services, benefits that people obtain from natural and semi-natural ecosystems, is inherently anthropogenic: it is the presence of human beings as valuing agents that enables translation of ecological structures and processes into value-laden entities (de Groot et al. 2002; Kremen and Ostfeld 2005). According to Slootweg et al. (2006), 'ecosystem services represent values of society'. The ecosystem approach principles set forth in the Convention on Biological Diversity (CBD 1999, 2004; Slootweg 2005; Treweek et al. 2005) align biodiversity and ecosystem services to relevant spatial and temporal scales by emphasising that management of biodiversity is a societal choice that includes stakeholder involvement and management of appropriate scale that takes into account spatial and temporal interconnections between biodiversity components, structures, and ecosystem processes, thus assessing and managing them in an integrated manner, not constrained by artificial boundaries.

Biodiversity can be placed in a spatial setting through the concept of green infrastructure. This concept was introduced in the USA in the late 1990s (Benedict and McMahon 2002, 2006). Benedict and McMahon (2006) define green infrastructure as 'an interconnected network of natural areas and other open spaces

that conserves natural ecosystem functions, sustains clear water and provides a wide array of benefits to people and wildlife'. Davies et al. (2006) define it, from a European perspective, as 'the physical environment within and between our cities, towns and villages. It is a network of multi-functional open spaces, including formal parks, gardens, woodlands, green corridors, waterways, street trees and open countryside. It comprises all environmental resources, and thus a green infrastructure approach also contributes towards sustainable resource management'. The European interpretation of green infrastructure is related to a fine-scale urban application wherein hybrid instruments of green spaces and built systems are planned and designed to support multiple ecosystem services (Pauleit et al. 2011). The green infrastructure emphasises both quality and quantity of urban, peri-urban, and rural interconnections; multi-functionality; and connectivity (van der Ryn and Cowan 1996; Turner 1996; Rudlin and Falk 1999; Schrijnen 2000; Benedict and McMahon 2002, 2006). The elements of green infrastructure have been seen as preserving and enhancing diversity within ecosystems in terms of habitats, species, and genes and as contributing ecosystem resilience (Ahern 2007; Tzoulas et al. 2007). The concept of the ecological network signifies much the same thing. The definition of an ecological network, according to Bennet and Witt (2001), is 'a coherent system of natural and semi-natural landscape elements that is configured and managed with the objective of maintaining and restoring ecological functions a means to conserve biodiversity while also providing appropriate opportunities for the sustainable use of resources'. The concept of greenways espoused by Ahern (2002), according to which a greenway system or network includes linear corridors and large areas of protected land that are physically and functionally connected, is very similar, but Opdam et al. (2006) see greenways exclusively as linear elements for multipurpose use, including nature conservation and aesthetics and also recreational and cultural purposes, while an ecological network is based more on coherence of ecological processes. They also stress flexibility

as a key feature of an ecological network, since the network can have different configurations and still serve the same goal.

According to Walmsley (2006) and Benedict and McMahon (2006), green infrastructure implies something we must have, in the form of a life-support system, in contrast to green space as merely something that is nice to have. The idea of infrastructure emphasises the interconnection of natural systems as opposed to separate parks and recreation sites (Walmsley 2006). It is not something stable that will take care of itself; instead, it requires proactive planning as a coherent entity (Sandstöm 2002). In addition, through the term ‘infrastructure’, green-space planning is aligned and put on a par with other infrastructures, such as transport, communication, water supply, and wastewater systems, and green spaces and overall built-up structure are, accordingly, viewed as integrated (Pauleit et al. 2011). Green-space planning comprises as well

an idea of communicative and socially inclusive management in the form of collaboration and mutual understanding by planners, the public, and decision-makers with respect to the benefits and losses entailed by different land-use options (Opdam et al. 2006; Pauleit et al. 2011).

It can be said in summary that aspects of composition, structure, and key processes represent the ecological conceptualisation of biodiversity; the green infrastructure is a spatial representation of biodiversity creating benefits for humans. Ecosystem services represent the socio-economic valuation of these benefits (see *Article IV*, Figure 3). The relationships among biodiversity, ecosystem services, and green infrastructure in environmental assessment and ecological impact assessment are illustrated in Figure 2.

Both environmental assessment and spatial planning deal with allocation of space in complex situations characterised by uncertainty and

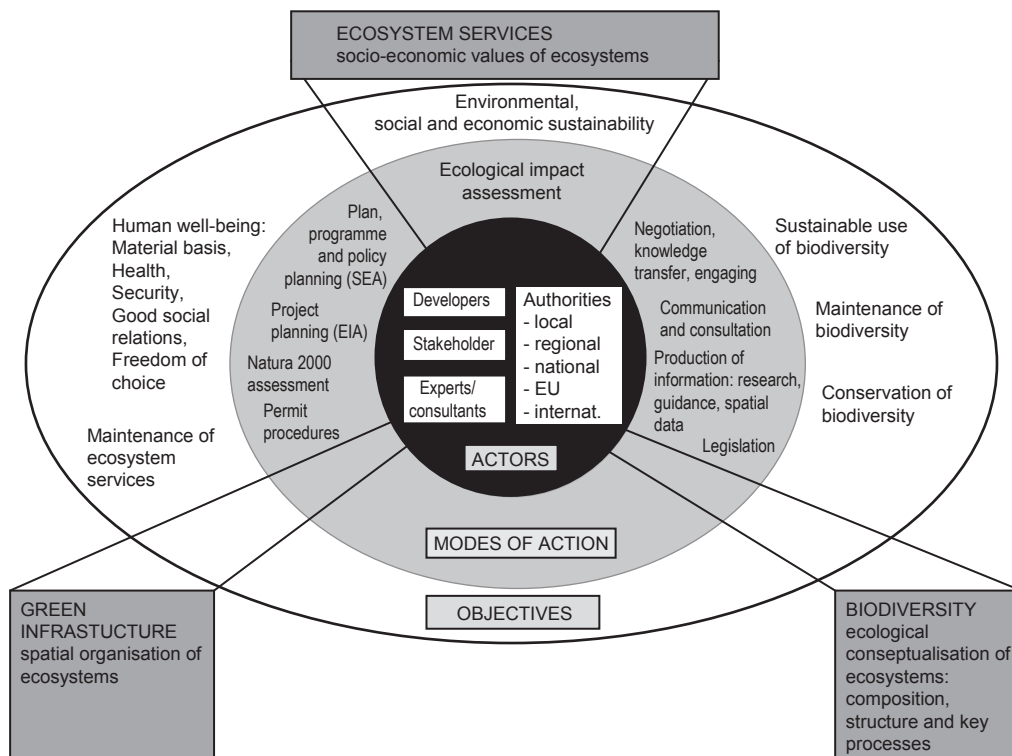


Figure 2. Biodiversity, ecosystem services and green infrastructure in environmental assessment.

conflicting values. According to Faludi and van der Valk (1994), planning is about resolving spatial conflicts between different land-use interests under conditions of uncertainty. Hillier (2010) defines spatial planning as a ‘perspective which draws out the spatial dimensions how to think about deliberate efforts to manage and develop place qualities and to pay attention to spatial connectivities’. Environmental assessment and its best-practice principles have been produced in parallel with planning theory in the field of spatial planning in recent decades, though along separate paths (Lawrence 2000). To understand the development of environmental assessment, it is helpful to consider environmental assessment in terms of the substantive (i.e., concerned with the substance of what the planning field deals with) and procedural (concerned with the processes of planning) approaches and conceptual challenges of planning (Kørnøv and Thissen 2000; Lawrence 2000; Benson 2003; Wilkins 2003; Weston 2003, 2010; Connelly and Richardson 2005; Richardson 2005; Isaksson et al. 2009). The Greek word ‘*theoria*’ refers to visual sight; appropriately, Forester (2004) uses the analogy of a telescope through which we can look at an issue for planning theory. However, there is no consensus as to any single view of planning theory. Richardson (2005) argues that planning theory cannot be organised into separate typologies for transfer of their synthesis to environmental assessment practice. Furthermore, it has been argued that a planning theory does not exist as a distinct theoretical sphere or autonomous body and instead consists of a wide range of parallel, incompatible, and competing theories and theoretical references from social and political science, decision-making, economics, psychology, geography, art history, aesthetics, etc. (Bengs 2005). Additionally, planning often addresses situations wherein more than one theoretical or normative approach is of relevance (Forester 1989; Taylor 1998; Hillier 2010).

Discussion of relations of planning theories and environmental assessment in the EIA and SEA literature has mostly concentrated on criticism of instrumental rationalist planning and on information production and deliberative ap-

proaches that emphasise dialogue and social learning as a replacement for rationalistic planning (e.g., Lawrence 2000; Elling 2004, 2009; Bjarnadóttir 2008; Weston 2010).

3.2 Information, contrasted to knowledge, and its use and impact in decision-making

Environmental assessment can be traced back to the instrumental rationalistic approach to planning and decision-making, which prevailed in the 1960s. This required technical evaluation to provide an objective basis for improved decision-making (Lawrence 2000; Weston 2000; Owens et al. 2004). The ideal of the technical-rational planning process was a simple one: survey, analyse, and plan. The rational planning process includes also a problem, need, or opportunity to be addressed; goals, objectives, and criteria; the generation and evaluation of alternatives; and explicit links to implementation (Lawrence 2000).

This technical-rational model has been applied in many assessment tools (Petts 1999) for decision-aiding and decision-making, and environmental assessment is one of them. The objective in rational-technical environmental assessment is provision of ‘value-free’ information about the affected environment (Bjarnadóttir 2008). After this, positive and negative effects of the chosen alternatives related to the initiative are balanced with the information acquired for the environmental assessment report, so that a decision on the optimal situation for the affected environment can be made for the implementation (Elling 2009).

The evaluation of whether environmental assessment results in the kinds of outcomes that are typically sought has been typically expressed in terms of ‘effectiveness’ (Jay et al. 2007). Analysis of effectiveness is intended to determine how much difference environmental assessment is making. It can be applied to consideration of changes in environmental quality that are very difficult to trace as results of individual assessments (Jay et al. 2007) or for verifying performance – ensuring that environ-

mental considerations are taken into account in decision-making (Glasson et al. 1999).

Wood and Jones (1997) found in their study of effectiveness nearly 15 years ago that, of 40 cases of EIA, only one had had significant influence on the result of decision-making, in one case where development was permitted, and that environmental impact assessment reports had played a significant role in only a minority of cases. EIA was seen as a process external to decision-making and had only a minor or, at most, moderate, fine-tuning effect on decisions concerning the projects. The contribution of EIA to project decisions has been very limited, and it is common that findings of EIA are marginalised in favour of non-environment-related objectives and political factors (Wood 2003; Cashmore et al. 2004).

These findings are consistent with the criticisms of EIA – and also SEA – as a technical-rational approach to decision-making (Jay et al. 2007). In general, a scientific, positivistic approach will not be generally appropriate for the messy problems often encountered in environmental assessment, which often cut across boundaries between scientific disciplines (Lawrence 1997). Instead of ‘value-free’ objectivity, decision-making is intricate interviewing of facts and values (Owens et al. 2004). Decisions are based upon values and interests of decision-makers operating within a political arena (Owens et al. 2004). Decisions are not made according to the logic of the technical-rational model; instead, they are influenced by ‘non-scientific’ factors, such as agency and corporate power and interest-group politics. Decisions are determined more by the goals of proponents or authorities and politics than by scientific impact studies (Lawrence 2000). Jay et al. (2007) argue that even if the environmental report presents environmental information satisfactorily – i.e., performs well – it is unlikely to succeed in its stated aim of ensuring that environmental considerations are fully incorporated into the decision-making.

Besides instrumental rationality, environmental assessment builds strongly on communicative strands of planning theory from the 1990s. This has been reflected in an upsurge of col-

laborative theory and practice in environmental assessment in 2000s environmental assessment development, in response to the weaknesses of environmental assessment that stem from instrumental rationalistic approaches (Richardson 2005). Communication and collaboration planning theory is based on planning theories that criticised rationalism and builds on communications theory (Forester 1989; Habermas 1984) and public participation. This theory focuses on consensus-building; accordingly, planning should occur through group deliberation, free discussion of argumentation, and negotiation. However, consensus-building approaches do not mesh well with resistance to change, highly complex issues, and large-scale and long-term planning situations wherein not all affected parties can be involved (Lawrence 2000).

The collaborative and communicative approach has not been able to resolve how to deal with the presence of multiple, often conflicting values and ways of assigning value in environmental assessment (Richardson 2005). Values have been interpreted to be ‘beliefs, either individual or social, about what is important in life’ (RCEP 1999, in Wilkins 2003). These can be expressed in economic, social, and ecological terms in environmental assessment (Slootweg 2005). There are many definitions of values and traditions of different disciplines and management systems, along with various methods of valuation, linked to use and non-use values of biodiversity, which may be economic or non-economic, intrinsic, existence values, cultural values, functional values, and/or research and education values (Erikstad et al. 2007; Wale and Yalew 2010). In particular, techniques for monetising the value of biodiversity in environmental assessment have inherent difficulties, with the result being little more than an indication of monetary value based on many approximations and aggregations (Wale and Yalew 2010). Similarly, monetising ecosystem services that are already representations of what is subjectively considered valuable is challenging (TEEB 2008, 2010; Kumar 2010; ten Brink 2011).

Environmental assessment is an element in a process in which actors – planners, politicians,

and stakeholders – persuade, mediate, and contest diverging interests and values (Runhaar 2009). Daniels and Walker (1996) argue that environmental assessment should provide a political setting for value differences to be mediated through decisions and settlement of conflicts. By contrast, Elling (2004, 2009) argues that environmental assessment is an arena of deliberation between different opinions, values, and interests, with no attempt at mediation or settlement. He argues that solving planning problems is left to the politicians, whose judgements and trade-offs are informed by the outputs of environmental assessment (Elling 2004, 2009). According to the instrumental-rationalistic model, the values and targets are set beforehand by developers, authorities, and science (Elling 2009). In a communicative-deliberative model, values and interests of stakeholders (including also ethical and aesthetic aspects) are brought into a public debate without any predefined objective but with a common objective – sometimes giving room for the original proposal and sometimes not (Elling 2009).

Especially in EIA, the assessment has been undertaken under the assumption that there is only one major decision, at a single point in time (usually connected with the report) where the results of the assessment are considered by those responsible for planning (Beanlands 1988). However, it seems naïve to believe that environmental assessment is solely decision-informing, or rational or ‘value-free’; it is certainly often decision-forcing if not decision-making (Benson 2003). Wilkins (2003) and Richardson (2005) see environmental assessment and values of developers/proponents, regulators/authorities, and the public as inextricably linked and integrated because of the reality of environmental assessment activity, which involves constant subjective micro-level judgements, from screening of proposals to final decision-making, that cannot help but deal with questions of value. Assessors’ personal values and subjective choices determine the methodologies and environmental considerations that are inputs to the assessment process (Morgan 1988; Wilkins 2003) and even more to interpreting, predicting impacts, and evaluat-

ing their significance (Lawrence 1993; Beattie 1995). Values are crucial even in construction of environmental assessment frameworks and tools (Richardson 2005; Bjarnadóttir 2008). As a consequence, a separate value assessment (e.g., for biodiversity aspects) or value criteria are not ‘value-free’ tools for valuation by stakeholders but already influenced by many subjective choices made in development of the assessment tools.

Wilkins (2003) and Richardson (2005) see environmental assessment as political to its core and the interplay of power and value as inescapable at every step in it. They both believe that the mediation of values is a constant feature of environmental assessment and it should be seen as a system or forum producing knowledge and as a source for directing the development of social values – if not changing, at least challenging interests of individuals linked with the interests of other people, not only as a means of making informed (or evidence-based) decisions. This is often referred to as social learning in EIA (Wandesforde-Smith and Kerbavaz 1988). Thus the values of proponents, authorities, and the public are shaped throughout the process of environmental assessment, through deliberative discourse and constant choices made in the assessment process. Wallington et al. (2007) call this a transformative environmental assessment approach that seeks lessons from policy-related disciplines and is intentionally both political and aimed at contributing to longer-term changes in values, worldviews, behaviours, and practices of actors and institutions.

According to Richardson (2005), there is no single approach that could discriminate facts from opinions, provide comprehensive knowledge, and eliminate the possibility of bias or distortions by politics. Therefore, combination of approaches would be most suitable and environmental assessment can be understood as a field of practice within which difficult choices are made about questions of value (Richardson 2005). These choices are made on the basis of both scientific analysis (due to changes in land use and biophysical environment affecting the biodiversity and ecosystem services it creates)

and open public deliberation (due to different valuation of ecosystem services), so knowledge is contested and shaped throughout the whole process of environmental assessment. Accordingly, instead of environmental assessment making minor changes to a proposal as an output of a rational or communicative approach, there should be a process wherein the whole proposal, its environmental objectives, alternatives, and knowledge should be shaped.

In this sense, environmental assessment can be used as a platform of knowledge brokerage including issues of communication, interaction, sharing of knowledge, learning, and contribution to common understanding, as well as effective action (Fischer et al. 2009; Sheate and Partidário 2010). The difference between 'value-free' information and knowledge is that knowledge implies that the information processed through learning can be recalled and so create understanding and insight, according to Sheate and Partidário (2010). Building on the work of Ward et al. (2009), they summarise three categories of knowledge-sharing:

- i. informing: knowledge management in the form of relatively passive dissemination of knowledge,
- ii. engaging: active linkage, collaboration, and exchange of knowledge among actors, and
- iii. building capacity: fostering greater self-reliance on the part of all actors – e.g., enhancing actors' knowledge transfer/communication skills.

These authors suggest that knowledge brokerage interpreted within the two last categories in environmental impact assessment offers the opportunity to move assessment techniques beyond information provision and toward learning to facilitate sharing of different forms of knowledge by using different techniques, such as interactive and participatory stakeholder engagement, workshops, network analysis, and use of geographical information systems and mapping. The typology resembles Fischer's (2007) categories of involvement of the public, stakeholders, and interested parties for the

whole assessment process by the planners and assessors via

- i. communication: a one-way process in which the objective is to inform third parties and the public and to assist them toward understanding of problems, alternatives, opportunities, and solutions,
- ii. consultation: a process of engagement in which external persons (for example, the public) are called to comment on documentation, and
- iii. participation: engagement in which external parties (such as the public) are called to contribute to the decision-making process by exchanging information, predictions, opinions, interests, and values.

What is the difference between EIA and SEA knowledge brokerage? Elling (2009) pinpoints it by stating that a project is something definite that describes specific actions proposed for implementation and is, once implemented, a reality for many years. A plan is something dynamic and describes intentions for the regulation of future activities; if the plan does not function as intended, it can be changed, as can the assessment of its environmental impact. EIA can provide moments for knowledge brokerage or deliberation (Isaksson et al. 2009) that are very important for the outcome of an individual environmental assessment and provide early-phase insights into potential conflicts of interests that depend on differences in valuation of ecosystem services (Slootweg and Kolhoff 2003). Monitoring can also serve as a means of learning and reshaping of the assessment and ultimately of the project itself. However, changes in values do not occur overnight, and they require continual discourse if they are to develop and evolve beyond the short time span of an EIA (Wilkins 2003). Thus SEA as an iterative tool linked with a repeated planning cycle offers fuller possibilities for knowledge brokerage.

3.3 Ecological impact assessment procedure

3.3.1 The process and its key issues

The procedural character is an obvious feature of environmental assessment. In the environmental assessment literature, the assessment process has been divided into several – though not necessarily strictly sequential – phases: screening, scoping, baseline studies, impact prediction and evaluation, mitigation, review, and monitoring. Sometimes formal decision-making is separated out as a clear phase, but, as illustrated in Section 3.2 of this work, decision-making is connected to every phase of the assessment process. In general, the idea that all environmental assessments have the same procedural steps that can be given specific names is a simplification. In reality, in planning systems, the assessments are messy, with unclear system boundaries, and their procedural stages cannot be clearly distinguished from each other (e.g., Hildén 2000; Kørnøv and Thiessen 2000). However, dividing the assessment process into several phases while mindful that these phases are iterative and overlap in real planning and decision-making situations is helpful in addressing procedural and substantive content of ecological impact assessment.

The EIA process is considered more rigorous than the SEA process, which can have more variations and flexibility and whose process and approaches must be developed and tailored to the institutional, political, and planning settings (Dalal-Clayton and Sadler 1999; Partidário 1999; Verheem and Tonk 2000; Dusik and Sadler, 2004). Vicente and Partidário (2006) argue that SEA should be not a streamlined sequence of standard activities but, rather, a framework for activities that enable SEA to be flexible, adaptable, diversified, and tailor-made for the decision-making process. However, flexible and adaptable should not mean vague and confusing (Retief 2007).

The treatment of biodiversity issues is an integral part of the environmental assessment process. This consideration of biodiversity in the environmental assessment process has been

called ecological impact assessment (Treweek et al. 1993; Treweek et al. 1998; Treweek 1999; Byron et al. 2000; Mandelik et al. 2005a), biological impact assessment (Atkinson 1985), biodiversity impact assessment (Byron 2000; Bagri et al. 1998), biodiversity(-inclusive) assessment (Sherrington 2005; Slootweg 2005; Gontier et al. 2006; Slootweg et al. 2006), or simply inclusion of biodiversity considerations in an environmental assessment (Hirsch 1993; Slootweg and Kolhoff 2003; Wegner et al. 2005). Treweek (1999) characterises ecological impact assessment as the process of identifying, quantifying, and evaluating the potential impacts of the specified actions on ecosystems or their components. Slootweg et al. (2006) emphasise valuation of ecosystem services provided by biodiversity. The general term ‘ecological impact assessment’ mirrors in my mind appropriately a combined/merged assessment process including both 1) focus of the environmental assessment on inevitably spatially bounded biophysical environment and biodiversity as composition, structure, and key processes and 2) an approach valuing biodiversity in terms of ecosystem services that require choices among several actors and stakeholders.

Key substantive and procedural issues of ecological impact assessment have been discussed in the environmental assessment literature (e.g., Treweek 1999; Byron 2000; Slootweg 2005; Slootweg et al. 2006). Flowcharts of various types have been produced to illustrate best procedural practice for ecological impact assessment (e.g., Atkinson et al. 2000; Byron 2000; Slootweg 2005). I will not present one, since the process charts are either too generic to address key issues of ecological impact assessment or, when detailed, too specific to be applicable beyond a single case.

There are certain fundamental issues attached to all phases of the procedure, and these affect approaches throughout the process of determining what is considered important and on what spatial and temporal scales, with what level of detail and uncertainty, and by whom. These issues have been discussed from the perspective of determination of impacts’ significance (e.g., Sadler 1996; Hildén 1997; Lawrence 2007a,

2007b, 2007c; Wood 2008) and cumulative impact assessment (e.g., Burris and Canter 1997; Piper 2001; Cooper and Sheate 2002; Therivel and Ross 2007; Canter and Ross 2010; Gunn and Noble 2011). Scholars have addressed them also as issues of scale (e.g., Gibson et al. 2000; João 2002, 2007a, 2007b; Partidário 2007; Therivel and Ross 2007; Moss and Newig 2010). In general, they have also been categorised as contextual issues (Marsden 1998; Fischer and Gazzola 2006; Hilding-Rydevik and Bjarnadóttir 2007; Runhaar and Driessen 2007; Runhaar 2009).

Determination of impact significance

Environmental assessment considers potential significant environmental impacts. This represents an initial attempt to narrow the scope of the assessment to the most important possible effects. It recognises that not all potential impacts can be considered, they cannot all be considered to the same level of detail, and impacts vary in their importance for decision-making (Lawrence 2007b). Determination of significance involves judgements about what is important, desirable, or acceptable (Sippe 1999) and is widely recognised as a vital and critical environmental assessment activity (Lawrence 2007b). However, any consideration of the significance of environmental effects must acknowledge that environmental assessment is inherently an anthropogenic concept (Beanlands 1988). As shown in Section 3.1 of this work, it ultimately involves society's value judgements surrounding the significance or importance of effects of human activity. These value judgements are often based on social and economic criteria (Beanlands 1988). The judgements reflect a political reality of impact assessment in which the significance is translated into public acceptability and desirability (Beanlands 1988). This value-bound nature of environmental impact assessment has made determination of impacts' significance one of the most critical aspects but at the same time the most complex and poorly understood, contentious aspect of the environmental assessment process (Duinker and Beanlands 1986; Sadler 1996; Wood 2008). The evaluation of

impact significance is a dynamic activity affecting choices in the assessment process in every phase: screening (deciding whether or not to make an environmental assessment); scoping (deciding what impacts and of which options to consider, along with which data, from where, and acquired through which methods); impact prediction, evaluation, and mitigation (deciding what impacts are judged to be significant and in need of mitigation and by which criteria); review (deciding on the adequacy of handling of impacts and what residual impacts are still regarded as too severe); and monitoring (deciding what impacts are worth monitoring) (Hildén 1997; Lawrence 2007a; Wood 2008).

Determination of significance is connected to the theoretical foundations for the environmental assessment. Lawrence (2007a) differentiates among three procedural approaches in the quest for significance: a technical approach integrating technical and scientific analyses into impact assessment; a collaborative approach incorporating community knowledge and perspectives; and the reasoned argumentation approach, which is effective in deriving and documenting the rationale from several sources for significance judgements in a form that all actors can understand or potentially support. There are also variations involving a composite of the three approaches, which in ideal form offer potential to link and combine technical analysis/knowledge with community knowledge/perspectives and qualitative data with quantitative; combine objectives, analysis, and values; combine multiple forms of expression (e.g., written, visual-aid, and oral); generate solutions and insights wherein the whole is more than the sum of its parts; and bridge the various actors' perspectives, interests, and values (Lawrence 2007a). However, Lawrence (2007a) points out that composite approaches can, if poorly designed and applied, be costly, difficult to understand, and time-consuming, and sometimes it is impossible to reconcile or counterbalance fundamentally different value-based perspectives on what is important and why. In addition, he argues that it may be better on some occasions to take a hard line on what is important for substantive environmental rea-

sons rather than adopt a composite significance determination approach, as the latter may lead to unnecessary environmental impacts or compromises in the quest for consensus. The substantive reasons for significance determination may be found in certain broadly acknowledged principles of international treaties (Pritchard 2005); guiding principles of best approaches (IAIA 2004), including the principle of 'no net loss', an ecosystem approach, sustainable use, equitable sharing, the precautionary principle, and a participatory approach; national and international legislation and policies (Slootweg et al. 2006); or other principles chosen for application in a particular impact assessment (e.g., net positive impact, net public benefit, definition of threshold levels, sustainability, or local and regional communities and environment net beneficiaries) (Lawrence 2007b). Depending on the planning and decision-making situation, it is essential, in SEA especially, to include positive impacts in determination of significance. Bringing both positive and negative impacts into the assessment enables comparison and trade-offs between negative and positive impacts when possible and consideration of the distribution of benefits over space, time, population groups and sectors of society, and affected receptors considered to be important (Lawrence 2007c). Since the significance determinations guide the whole assessment process and its content, decisions on impact significance should be made as early as possible in the environmental assessment process – during scoping, at the latest. According to Lawrence (2007a), such decisions should be explicit, substantiated, and collaborative and should involve interested and affected parties.

Assessment of cumulative effects

Cumulative effect assessment explores whether individual insignificant impacts become significant when combined, at the level of the initiative and in conjunction with past, present, or likely future activities affecting the same environment (Lawrence 2007c). Treweek et al. (2005) define cumulative effects as effects occurring when thresholds for stability or viability, prevention of sudden decline, or collapse in

biodiversity are exceeded, causing biodiversity decline that cannot be attributed to any single action. Because of the great interconnectedness within and between ecosystems, most biophysical changes result in a cascade-like chain of events (Slootweg and Kolhoff 2003). Thus impacts on biodiversity and ecosystem services are typically cumulative. Cumulative impact assessment does not concern the effects of a particular project, plan, programme, or policy; it cuts in the opposite direction, focusing instead on the receiving environment (Therivel and Ross 2007; Canter and Ross 2010; Gunn and Noble 2011). In the case of ecological impact assessment, the receiving environment means biodiversity elements, by which I refer to individual features of the biodiversity aspects of composition, structure, and key processes. Only the Natura 2000 appropriate assessment process has the receiving environment focus as a starting point already in the screening phase of the project, specifying that 'any plan or project [...] likely to have significant effect thereon, either individually or in combination with other plans or projects[,] shall be subject to appropriate assessment' (CEC 1992; European Commission 2000, 2001). While EIA and SEA directives require describing cumulative impacts only as an output of impact assessment (CEC 1997, Annex IV; CEC 2001, Annex I), unlike the wording of the Habitats Directive, the description of the Natura 2000 process indicates that cumulative impacts are a result of many activities, which may not have been caused by any specific plans or projects and may have built up over time via numerous inter-linked actions (e.g., climate change) (Therivel and Ross 2007).

The cross-cutting across time and space, differences in planning and decision-making processes and in their actors and stakeholders, and linkage to other past or future activities (of any type, not necessarily connected to EIA and SEA procedures) make cumulative impact assessment extremely challenging. It is regarded as one of the most persistent challenges in environmental assessment (Gunn and Noble 2011). Individual EIAs have systematically failed to address cumulative effects (Burris and Canter

1997; Atkinson et al. 2001; Piper 2001; Cooper and Sheate 2002; Bismar 2004).

The main steps of cumulative effect assessment are rather straightforward:

1. identify the affected receptors,
2. determine what past, present, and future human activities have affected or will affect these receptors, and what has led to these activities,
3. predict the effects of the project/plan on the receptors, in combination with the effects of other human activities, and determine the significance of the effects, and
4. suggest how to manage the cumulative effects (Ross 1998).

The exercise becomes complicated when one considers the level of planning and decision-making at which the cumulative impact assessment should be undertaken: project level or plan, programme, or even policy level? Sonntag et al. (1987) argue that cumulative effects on a regional scale can be controlled only through planning processes directing development at that scale. Furthermore, Noble (2008) sees properly assessing and managing cumulative impacts as beyond the scope and scale of project-based EIA. Treweek (1999) argues that failure to deal effectively with cumulative ecological impacts is one of the main arguments for strategic environmental assessment. Indeed, among the intended main benefits of SEA was that it should allow for better consideration of cumulative effects than project EIA does (Fischer 2002; Therivel 2004). However, some scholars are not convinced of SEA's ability to deal with cumulative effects. Gunn and Noble (2011) point out that the anticipated benefits of assessing cumulative effects in SEA are well documented but there are few practical examples that demonstrate these benefits. In particular, very strategic-level SEAs do not focus on impact assessment and instead are used as an aid in objective-setting and evaluation (Partidário 2007; Gunn and Noble 2011).

Several challenges arise in connection with cumulative impact assessment. The first is linked to affected receptors related to the above discussion of significance. It is a comprehen-

sive exercise to decide which biodiversity elements are under consideration if these are not explicitly defined – for example, in a legislative framework such as the conservation objectives in the Habitats Directive (CEC 1992).

The second challenge is related to impact prediction. What human activities should be included in the assessment, and on what level of detail? How many similar projects and possible higher-tier plans and their 'inherited' predictions and other activities must be assessed that underlie trends and their impacts but are not included in specific plans or projects? Precise predictions would require use of complex modelling tools to acquire information on space-time lag, path dependencies, non-linear relationships, and positive and negative feedback mechanisms (Therivel and Ross 2007).

It is often impossible to measure the biodiversity consequences of human activities precisely or to predict them (Slootweg 2005). Cumulative impacts are especially difficult to predict accurately, but often assessors, usually consultants, are reluctant to produce cumulative effect predictions that are not very detailed, even when broad-brush assessment would suffice (Therivel and Ross 2007). Slootweg and Kolhoff (2003) and Slootweg (2010) also see that detailed quantified information on biodiversity sometimes is not necessary and it is possible to make good qualitative judgements on biophysical changes in ecosystems without, for example, detailed knowledge of ecosystems' species composition and abundance. They recognise that an experienced ecologist will be able to make comparative statements on the magnitude of the impacts when comparing the alternative options of the initiative and thus provide relevant information on the expected impacts on biodiversity, without having to go into detail. The reluctance to produce general-level impact predictions is partly due to the possibility of reviewers challenging superficial predictions and demanding greater detail and partly because the impact-assessors may see themselves as making sound scientific predictions and do not wish to put their reputation at risk with anything less (Therivel and Ross 2007). However, the avoidance of general-level

predictions may leave important aspects outside the decision-making. Therefore, broad-brush prediction is better than no prediction at all (Therivel and Ross 2007).

Furthermore, assumptions and uncertainty are present in all impact assessments, especially in cumulative assessment, where long time horizons – of several decades – are considered. However, consideration of uncertainty is generally neglected in environmental impact assessment (Glasson et al. 1999; Benson 2003). Impact prediction dealing with biodiversity involves an especially large extent of simplification and uncertainty linked to the data (temporal and regional coverage, relevance, and accuracy), the methodologies used (assumptions made, methods and tools chosen, and boundaries defined), and value judgements provided by the experts and other actors involved (rarity, vulnerability, and user values) (Southerland 1995; Treweek 1996; de Jongh 1998; Geneletti 2002; Geneletti et al. 2003). This is due to the complexity of the ecosystems and of the interactions among and/or between populations, species, and biotic and abiotic processes at spatial and temporal scales and makes it extremely difficult to adapt a holistic management framework (Erikstad et al. 2007). De Jongh (1998) proposes the use both of socio-scientific methods to focus on subjective elements and definition of uncertainties and of techno-scientific management tools to reduce the uncertainty of impact predictions. Geneletti (2002) proposes specific but simple uncertainty analyses as an integral part of ecological impact assessment to support decision-making by making difficulties and uncertainties explicit. Partidário (2007) emphasises the importance of problem definition. Lee (2006) defines problems as unmet goals. Partidário (2007) argues that weak or deficient analysis often results more from bad definition of the problem (Levitt and Dubner 2005) than lack of data. Thus an avalanche of data to overcome uncertainty can disturb the focus and determination of the broad perspective needed for understanding of the whole planning situation and the biodiversity elements and ecosystem services at stake. Therefore, also inaccurate and general

predictions can be considered valid as long as uncertainties are made explicit and transparent.

The third challenge is linked to the measures for management and mitigation of cumulative effects. They require cumulative actions: the concerted action of various actors and stakeholders on several spatial levels between proponents, planners, authorities, and multiple stakeholders (Therivel and Ross 2007; Canter and Ross 2010). On project level, consent regimes can set certain conditions for mitigation and management, whilst management of cumulative effects will be voluntary at the plan or programme level – unless standards or thresholds have been externally imposed – so less likely to occur (Therivel and Ross 2007). The management of cumulative effects is thus dependent on the spatial scale of planning and decision-making. On plan level, management and mitigation measures are multiple but mostly without formal standards. At plan and programme level, these measures include not only project-level ones (e.g., requiring a given type of management for each project) but also location-associated measures (allowing projects here but not there), cross-project measures (e.g., individual developers' contribution to a fund to reach a management goal), demand-reduction and other measures to promote behavioural change by individuals (e.g., congestion charges), and other strategic measures (e.g., related to building density). That most plans or programmes have a greater physical extent allows scale-based measures to be put in place that are infeasible for most projects (e.g., new parks to serve multiple housing projects). The greater temporal extent of plans and programmes allows for time-related management measures (e.g., X cannot be built until Y is in place) (Therivel and Ross 2007).

Scale dimensions and integration into decision-making

Partidário (2007) uses scale in environmental assessment to mean the extent of spatial assessment or the time period considered, with the extent determining the size of the 'window' for viewing the world (Goodchild and Quattorchi 1997). João (2007a) identifies two key mean-

ings of scale: spatial extent (e.g., the size of the area studied) and the level of detail or granularity used (e.g., sampling rate). Both can be applied on temporal as well as spatial scale. Scale issues have been considered in other ways, such as with respect to the timeliness of the assessment – i.e., integration with the temporal scale of decision-making: the decisional time scale, which may be measured in days, weeks, or months (Partidário 2007). If impact assessment cannot be achieved within reasonable time, the window of opportunity for its widespread use is likely to close (Lee 2006). Scale could comprise, in addition to spatial and temporal dimensions, an analytical dimension used to study and measure a specific phenomenon or observable fact (Gibson et al. 2000). This could cover the different procedural and substantive approaches in determination of significance and different theoretical approaches taken in environmental impact assessment. When analytical dimensions are considered, scalar dimension has been defined as an analytical dimension of the problem under study, with scalar level being a particular level on a scalar dimension (Moss and Newig 2010). For example, if the impact assessment looks at the dimension of compositional biodiversity, the levels would run from a gene to the global ecosystems. In examination of sustainability, the levels might be the social, environmental, and economic. Jurisdictional scale levels could be a Member State and the European Union or something else, depending on the problem at hand. Scale choices affect the results of environmental assessment. It is worth noting that scale can be abused, intentionally or unintentionally: actors may want to change the spatial scale to fit their objectives – for example, leaving important levels outside the assessment (e.g., biodiversity considerations) – or change the scale during the phases from screening to monitoring, depending on the scale at which the impacts might appear the least significant (Karstens et al. 2007). Strategic behaviour of actors in setting impact assessment boundaries could involve defining the study area to be so large that impacts will be ‘diluted’ to the larger spatial area (Ross 1998; João 2002); e.g., for a developer interested in getting the proposal

accepted, it is more beneficial to use a larger study area. Misuse can occur if the level of scale dimension used changes during the assessment process with raising of the threshold for consideration of an impact as significant (Wood 2008). This could happen when certain categories or criteria are chosen or created early in the assessment but the threshold levels are changed during the assessment to keep the impacts below the level deemed unacceptable.

Scale is connected also to the ‘tiering’ of the assessment, in what has also been described as ‘vertical integration’ or the ‘trickle-down effect’ (Therivel and Partidário 1996; Noble 2002). Tiering is about how different levels of planning relate to each other (Arts et al. 2005). Impact assessment is described as a tiered or layered process in which decisions on a higher level steer decision-making at a lower level. Tiering is often attached to the relations of SEA and EIA. In the ideal situation, the planning process starts with the policy-setting objectives and background for proposed actions, usually with a sector-oriented or geographical scope. Policy objectives are operationalised to an action plan, and further programme work and actual operation is done in projects. Ideally, knowledge about effects on the regional level, including standards and thresholds, should trickle down to the project level so as to avoid significant adverse environmental impacts (Noble and Storey 2001; Fischer 2003). In practice, processes are rarely so streamlined and there are enduring questions around tiering and the extent to which it occurs in practice (Nooteboom 2000; Noble 2002; Arts et al. 2005; Gunn and Noble 2011).

The concept of tiering is often based on the naïve assumption that planning is linear. In reality, plans and programmes do not always precede projects, and information on a higher level can be outdated or of so normative/strategic a nature that EIA cannot just continue to refine the information. Furthermore, planning decisions and their impacts are often generated at project level and ‘evaporate up’ to be managed at a higher level of planning instead of ‘trickling down’ top-down (Arts et al. 2005). The spatial levels of planning and decision-making are inter-linked. Sometimes a solution for a problem

does not lie at the level at which the problem was defined (Partidário 2007). Therefore, collaboration and iterative assessment, planning, and decision-making that cross scale boundaries are needed. For effective tiering, integration between spatial scales, and proper cumulative effects assessment considering biodiversity, the project-level impact assessment should be informed beforehand with what kind of information from the project level is needed for the region-level impact assessment. Similarly, in the region-level impact assessment, planning and decision-making should feed in to the project-level impact assessment the assessment outcomes and how they can be used and taken into account in project-level impact assessment. The tiering mechanisms must then be built in the assessment process (Gunn and Noble 2011).

Scale aspects come under consideration in the linkage between assessment and decision-making. Slootweg et al. (2006) characterise three types of SEA + planning process combination. The first type is the parallel combination, wherein the environmental assessment and decision-making processes proceed simultaneously but separately and the assessment is intended to support the decision-making at the end of the process. The second type is the integrated combination, where the relevant assessment findings are input to the planning during key stages of the planning process (Lee 2006). These key stages in the planning process have also been called decision windows (Caratti et al. 2004) or integration moments (Partidário 2005). The third type is the environmental-assessment-led process, wherein the assessment creates the structure for the planning process in cases of an absent or weak planning process. The parallel process has been the most traditional approach for EIA-type SEAs. In the worst case, environmental assessment has been started at the end of the process as an add-on without much effect on the final decisions. In my mind, these combinations fit project-level assessment as well, since EIA is carried out to support design of projects and cannot be categorised as ideally a parallel process. I agree with Slootweg et al. (2006) as to the basic characterisation of linkages but add the micro-level value-bound decision-making (choices

dealing with alternatives, methods, etc.) made by assessors themselves or in collaboration with planners, authorities, and stakeholders throughout the assessment process. I also would consider instead of an environmental-assessment-led process a totally merged process of decision-making process and environmental assessment in line with what Partidário (2007) suggests, in which the environmental assessment is adjusted to its decisional scales. This process is planning- and decision-making-centred (Figure 3).

The context of the assessment

The context or the context-dependence of the assessment, especially in the SEA literature (Marsden 1998; Fischer and Gazzola 2006; Hilding-Rydevik and Bjarnadóttir 2007; Runhaar and Driessen 2007; Runhaar 2009), has been used as a kind of umbrella term to refer to important dimensions needing consideration in the setting up of environmental assessment and its integration with decision-making. However, there is no consensus on what constitutes a context (Runhaar 2009).

Hilding-Rydevik and Bjarnadóttir (2007) define context as the set of facts or circumstances that have an impact on the approaches to SEA chosen and the outcomes of SEA implementation. They consider contextual factors such as national policy style (top-down or interactive, open or closed), characteristics of the planning agency, planning style, and the extent of political commitment to sustainable development. Wood (2008) sees these as the institutional context and considers spatial scale, temporal change, social and ethical values, ecological sensitivity, economic considerations, etc. to be other context issues. Regarding SEA, Fischer and Gazzola (2006) consider to be context the various aspects of a sustainable development framework that provides for the aims and objectives underlying SEA, effective co-operation and public participation, and an effective EIA system with which SEA can be tiered. Slootweg and Kolhoff (2003) argue that an important consequence of context-dependency is that impacts on biodiversity cannot be determined by external experts only; one must consult the stakeholders who make use of the ecosystem services.

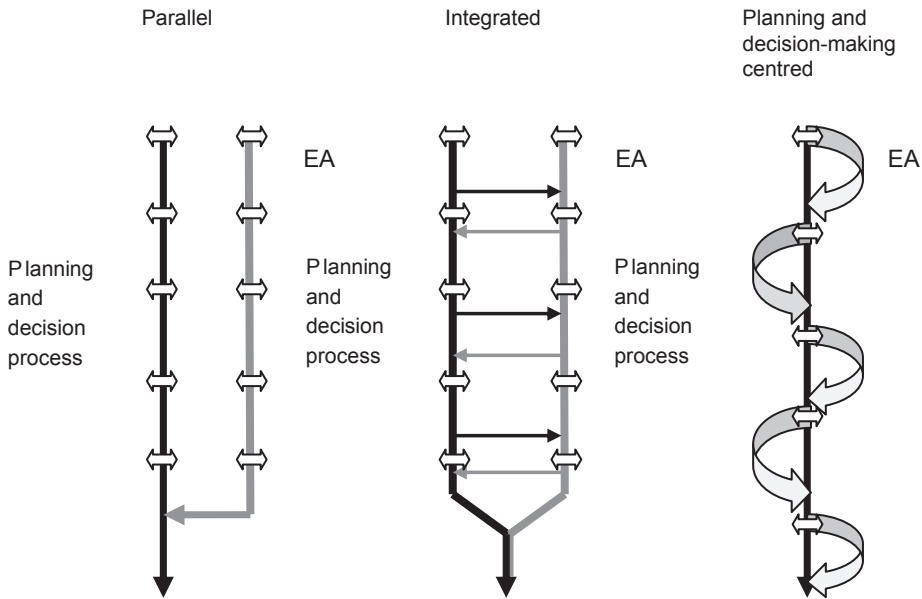


Figure 3. Linkage between environmental assessment, planning and decision-making. Small white arrows illustrate the micro-level decision-making done throughout the process regardless of the final formal decisions made after the completion of the process. Modified from Slootweg et al. (2006) and Partidário (2007).

Lee (2006) groups practical information needed to understand the context into three types: the regulatory and institutional context, the characteristics of the initiative to be assessed, and the resources available for the completion of the assessment. These include, *inter alia*, formal phases related to legal requirements and actors for the process in the regulatory framework, timing of the assessment process in relation to the planning process, and the available data and expertise. This resembles earlier conceptual thoughts of Beanlands and Duinker (1983) and Atkinson (1985) related to 'boundaries'. They recognise administrative boundaries (time and space limitations imposed on the assessment for political, social, and economic reasons), project boundaries (the temporal and spatial scale over which the project extends, physical boundaries (time and space limitations imposed by natural input/output transportation mechanisms and the physical barriers affecting the system), ecological boundaries (time and space scales within which the natural system operates), and technical boundaries (time and space limitations imposed by our abilities to predict and measure

ecological changes). Wallington et al. (2007) take a more abstract approach to the contextual dimensions, rooting their angles of contextuality in the earlier work of policy sciences, and suggest that the relevant context dimensions include the degree to which definite substantive knowledge is present, the level of agreement on values, the degree of conflict of interests, the power distribution, the extent of trust among participants in the process, the clarity and strictness of the procedures, and the character of the planning process (political controversy).

It can be concluded that the main point of the discussion surrounding contextuality is that the assessment approach and methods should be adapted to the specific situation and circumstances and it is important to explore the planning and decision-making situation comprehensively before planning any given impact assessment process. Because of the conceptual confusion as to what constitutes a context, I avoid using the concept of context in this work outside the discussion presented above.

3.3.2 Procedural and substantive content of the phases of ecological impact assessment

Screening

Screening in ecological impact assessment determines which proposed initiatives need or do not need further ecological impact assessment and indicates the level of impact assessment required. Bagri et al. (1998) called for a preliminary rapid assessment to determine key impacts, their magnitude and significance, and their importance to decision-making in terms specific to biodiversity. The first question may be whether there are any specific legislative requirements for EIA or SEA on the basis of biodiversity – for example, lists of initiatives to which it is obligatory to apply an environmental assessment or screening criteria for a case-by-case screening decision. For example, EIA and SEA directives set forth screen-

Table 1: Best practices in screening (compiled from the work of Slootweg and Kolhoff 2003; IAIA 2004; Treweek et al. 2005; Slootweg et al. 2006; and Rajvanshi 2010)

-
- addressing aspects of composition, structure, and function (key processes) of biodiversity
 - addressing all levels of biodiversity in terms of each of the aspects
 - using biodiversity triggers, including
 - impacts on protected areas and areas that are not protected but are important for biodiversity
 - activities posing a particular threat to biodiversity
 - areas that provide important ecosystem services
 - interventions acting as direct or indirect drivers of change
 - change of the physical environment such as causes extinction or change in loss of habitats or ecosystems or linked to maximal sustainable yield or the maximum allowable level of disturbance of a biodiversity element
 - area of influence, ecosystem, and the types of land use being affected
 - considering the influence of the initiative in terms of sustainable development goals, environmental quality, and health
 - considering the probability, duration, frequency, and reversibility of effects; cumulative effects; the magnitude and spatial extent of the effects; and the value and vulnerability of the area likely to be affected
-

ing criteria linked to whether an initiative has impact on biodiversity (CEC 1997, Annex III; CEC 2001, Annex II). The Habitats Directive (CEC 1992) includes its own screening phase and criteria linked to the conservation values for Natura 2000 sites (European Commission 2001). Slootweg et al. (2006) argue that legal criteria may not guarantee that biodiversity will be taken into account; e.g., important screening criteria can be found in national biodiversity strategies and action plans. Table 1 presents the items that the literature has included in best practice for screening.

Scoping

Treweek (1999) defines scoping as ‘all about ecological impact assessment design’. She argues that the importance of scoping cannot be overemphasised: get it wrong and important ecological components and effects may be absent from the environmental assessment entirely or discovered only when it is too late to do anything about them. Too narrow a scope will exclude important issues, but too broad a scope makes the assessment superficial and unfocused or too complicated to handle if everything is treated in detail (Slootweg and Kolhoff 2003). Considerable resources may be used for irrelevant details, or overly general ‘broad-brushing’ of all impacts may receive the focus instead of significant ones (Sadler 1996; Wood et al. 2006). Slootweg and Kolhoff (2003) emphasise distinction between a conceptual, holistic ‘wish list for ecological scoping’ and what is actually practical. For example, with the current state of knowledge and resources, the feasibility of detailed genetic studies in environmental assessment is highly questionable (Mandelik et al. 2005b). However, it may still be possible to identify situations in which there is strong likelihood of genetic impoverishment or isolation occurring without making precise predictions (Treweek et al. 2005).

Scoping defines more closely the characterisations of the screening and establishes key issues for the assessment. Mandelik et al. (2005a) found that the scoping phase and its result, a scoping document, which is usually called ‘Terms of References’ (ToR) and sometimes

Table 2: Best practices in scoping (collected from Kennedy and Ross 1992; Morris 1995; Bagri et al. 1998; Glasson et al. 1999; Treweek 1999; Byron 2000; Slootweg and Kolhoff 2003; Mandelik et al. 2005b; Treweek et al. 2005; Lee 2006; Slootweg et al. 2006; and Rajvanshi 2010)

-
- definition of goals, targets, purposes, and how they are related to biodiversity for the initiative and problem definition
 - identification of legal requirements
 - definition of the biodiversity objectives in the area affected by the initiative, including relevant policies, programmes, and plans
 - definition of the temporal, spatial, and thematic limits/boundaries within which the assessment is undertaken (related to impact area and the time and types of impacts)
 - examination of the characteristics of the initiative and its activities
 - selection and examination of biodiversity elements and land-use characteristics considered to be important or valuable and that merit detailed consideration in the assessment process with respect to the proposed initiative, also called valued ecosystem components (VECs) (Beanlands and Duinker 1983; Treweek 1999) – eventually these may be a ‘mixed bag’ of species, habitats, and ecological and economic functions of ecosystems
 - examination of any anticipated trends in biodiversity in the absence of the proposal
 - identification, in consultation with the stakeholders, of the ecosystem services and the users of ecosystem services / people who depend on these ecosystem services
 - identification of potential interactions between the receiving environment and the initiative, including the biophysical changes (in soil, water, air, flora, and fauna) expected to result from proposed activities or induced by any socio-economic changes caused by the activity
 - preliminary screening of potential impacts and their categories, including direct, indirect, secondary, cumulative, short-term, medium- and long-term, permanent and temporary, positive, and negative, and the main implications for people who use ecosystem services
 - analysis of the opportunities and constraints for biodiversity
 - consideration of mitigation (including avoidance, minimisation/reduction, or compensation) and enhancement measures
 - identification of the studies and data needed to gather information to support decision-making
 - early site visits
 - determination of the level of detail for the various parts of the assessment
 - definition of the methods to be used in the assessment (including data collection, impact prediction, evaluation of impact significance, and participation)
 - specification of impact indicators for monitoring
 - identification of gaps in knowledge
 - accommodation of data gaps and uncertainties
 - the need for expertise and experts in the assessment
 - specification of alternatives (location, scale, siting or layout, or technology alternatives) to the proposed initiative for assessment and preliminary information on the above-mentioned requirements
 - planning of reporting
-

‘Guidelines’, were the most important factor for determining the quality of the ecological impact assessment. It has also been reported that project-type-specific scoping guidelines ignoring biodiversity aspects contribute to their exclusion from the impact prediction and evaluation (Swangiang et al. 2004). Scoping is completed in a relatively short time. In Table 2, the items that have been included in the literature as best practice for scoping are presented.

Slootweg et al. (2006) outline that in SEA the scoping methods consist of a combination of political agenda, stakeholder discussions, and expert judgements while in the EIA scop-

ing methods have to do with a combination of local issues and technical checklists. Byron (2000) and Bagri et al. (1998) emphasise the involvement of community members, regulatory authorities, decision-makers, and outside experts besides the assessment team in scoping of EIA. Treweek (1999) argues that some sort of field study or original study will be necessary in most cases because of insufficiency in up-to-date site-specific information. However, the availability of GIS data may reduce the need to implement site or field surveys and has made it considerably easier to develop regional approaches (Treweek 1996, 1999). Preliminary

surveys may be needed simply to establish whether the habitats and species are present, to derive suitable limits for more detailed studies to be undertaken later.

Bagri et al. (1998) highlight the critical importance of consideration of alternatives in the earliest possible phase for effective integration of biodiversity issues. Morris (1995) argues that the most important alternative is the ‘no action’ alternative, which functions as a baseline to which the effects of the project will be compared. Slootweg et al. (2006) continue emphasising the importance of definition of baseline conditions for evaluating significance of impacts and argue that these must be quantified whenever possible. They point out the dynamics, implying that present development and that expected if the proposed initiative is not implemented must be included in scoping. In practice, producing this baseline scenario has been

proved to be challenging (Wathern 1988; Wale and Yalaw 2010).

Baseline studies

Baseline studies characterise the affected biodiversity elements and their conditions or state in the absence of any proposed action. In Table 3, the items that have been included in the literature as among best practices for scoping are presented.

Since it is impossible to measure everything precisely, Slootweg (2005) emphasises identification of situations that may result in serious consequences for biodiversity and subsequent identification of aspects that need to be studied for preventing large amounts of data (such as species lists) from being gathered without necessarily containing relevant material. Whilst this should be planned well already in the scoping, it might be useful to check it during the base-

Table 3: Best practices in baseline studies (collected from the work of Wathern 1988; Treweek 1999; Morris 1995; Byron 2000; Slootweg 2005; Treweek et al. 2005; and Rajvanshi 2010)

-
- addressing how biodiversity is organised in time and space
 - situating the baseline study in the wider spatial setting of the relevant biogeographical area(s)
 - trying to assemble the whole picture across spatial scales
 - reflecting seasonality and variation over multi-year time scales
 - studying areas that are likely to be affected
 - studying only the relevant issues
 - focusing on ecosystem processes and services that are critical to the integrity of ecosystems and human well-being
 - reflecting planner and decision-makers' needs
 - making good use of existing information
 - involving stakeholders or consultees for relevant information
 - addressing various aspects and levels of biodiversity explicitly
 - addressing key functional relationships and interdependencies
 - covering a range of key species (e.g., characteristic and species susceptible to habitat fragmentation) instead of just rare and endangered ones
 - addressing why biodiversity is important and to whom
 - predicting how conditions would develop in the absence of the initiative
 - collecting relevant information on other initiatives and activities
 - undertaking new fieldwork for collecting data answering clearly defined questions
 - undertaking the fieldwork at the right time in relation to the optimal sampling or observation period for the species or ecosystem surveyed
 - involving professionals with skills in interpreting the data collected
 - describing the results of the baseline studies on maps
 - assessing the importance of biodiversity elements – e.g., using evaluation criteria
 - reporting the details of the survey methods and the times of sampling and observations
 - in the reporting, including the lists of species and other details as appendices
 - in the reporting, providing an assessment of the uncertainties attached to the methods and data and how they limit the impact predictions
 - presenting intelligible and non-expert conclusions
-

line studies. Wathern (1988) argues that there is a tendency to give too much weight to baseline studies early in the assessment process, with a possible result being that there is a great deal of information made available on the environmental setting of a particular initiative, but it may be irrelevant to the resolution of certain critical questions raised in later phases of the process. Wathern (1988) regarded performing baseline studies without clearly defined objectives and thus wasting time and money on superficial surveys of relevant information for decision-making as a universal problem of EIA in the late 1980s.

Impact prediction and evaluation

Impact prediction and assessment identify and predict impacts on selected biodiversity elements by comparing against a baseline. Table 4 presents the items that have been cited among best practices in impact prediction in the literature.

Geneletti (2002) emphasises the difference between impact prediction, which is done to identify the impacts, and evaluation, to evaluate or assess their relevance. Evaluation is the phase in the assessment process in which all the information is brought together and consideration is given to whether the impacts are socially acceptable or not – in other words, whether the adverse effects are significant (Glasson et al.

Table 4: Best practice in impact prediction (collected from works by Morris 1995; Bagri et al. 1998; Treweek 1999; Byron 2000; IAIA 2004; Sloomweg 2005; Treweek et al. 2005; Sloomweg et al. 2006; and Rajvanshi 2010)

-
- examination of impacts identified in the screening and scoping stage in further detail
 - identification of impacts at the ecosystem, species, and gene level
 - reflection of long-term ecosystem processes, including long-term and delayed effects
 - inclusion of all categories of impacts, including direct, indirect (also delayed), associated (e.g., impacts of a project in the form of necessary infrastructure), cumulative (time- and space-crowded impacts, such as habitat loss due to isolation), and synergistic (e.g., toxic effects of mixture of several pollutants)
 - determination of the duration and reversibility of impacts (permanent and temporary, including time of occurrence)
 - recognition that biodiversity is affected by cultural, social, economic, and biophysical factors
 - consideration of the full range of factors affecting biodiversity, including indirect and direct drivers of change
 - consideration of cumulative threats caused by other activities and initiatives
 - provision of insights into cause-to-effect chains
 - quantification of changes where possible
 - understanding of recovery mechanisms and the time required for recovery from impacts
 - focus on processes critical to human well-being and ecosystem services
 - identification of impacts on values and uses of biodiversity
 - identification of the environmental conditions required to conserve or promote biodiversity
 - indication of the legal provisions that guide the decision-making
 - consideration of impacts for which no legal provision applies
 - evaluation of the significance of the impacts before mitigation
 - evaluation of the significance of each impact in consideration of the evaluation criteria used
 - definition of threshold values or 'limits of acceptable changes' to distinguish between non-significant and significant impacts for decision-making
 - description of the impacts of alternatives with reference to the baseline situation
 - review and redesign of alternatives
 - ranking of alternatives
 - treatment of the uncertainty of ecological predictions
 - in the reporting, description of the prediction and evaluation methods and the significance criteria applied
 - presenting of the possibilities and available techniques to mitigate impacts on biodiversity
 - presentation of intelligible and non-expert conclusions
-

1999). There are many arguments for prediction being one of the weakest features of impact assessment (e.g., Benson 2003; Lee 2006). As many as 55% of the predictions have been reported to be inaccurate, uncertain, non-quantifiable, or not verifiable (Dipper et al. 1998; Wood et al. 2000) – in other words, non-auditable or impossible to monitor.

Mitigation

Mitigation is addressed to redressing significant adverse effects on biodiversity. Mitigation should take many forms, given the limited effectiveness of many ecological restoration measures; therefore, every effort should be made to avoid significant adverse impacts before resorting to other measures, using the avoid – reduce – compensate for – enhance sequence (Treweek et al. 2005). Especially in SEA, mitigation should be aimed at keeping options open and flexible, involving ‘no-regret’ options that deliver benefits exceeding their costs: win–win options that both contribute to meeting the objectives of the initiative and enhance biodiversity and that avoid decisions that will make it more difficult to improve biodiversity in the future (Treweek et al. 2005). Items that have been

Table 5: Best practice of mitigation (collected from works by Bagri et al. 1998; Treweek 1999; IAIA 2004; Slootweg 2005; Treweek et al. 2005; and Rajvanshi 2010)

-
- usage of a mitigation hierarchy from best to worst: avoidance (or prevention), reduction (or mitigation), and – as a last resort – compensation
 - inclusion of enhancement of biodiversity
 - inclusion of ‘no net loss’ and precautionary principles
 - identification of ‘no go’ or ‘no exploitation’ areas
 - inclusion of targeted mitigation measures – i.e., identifying which mitigation measures mitigate identified impacts
 - consideration of only those mitigation measures that can be achieved in practice
 - securing of adequate funding for mitigation and ensuring handling of the responsibilities
 - assessment of the effectiveness of mitigation
 - consideration of the effects of the mitigation itself
 - evaluation of the residual impacts and their significance after mitigation
-

cited in the literature as among best practices in mitigation are presented in Table 5, below.

Review

The assessment often includes a review to ensure that the report follows the ToR and standards of good practice. In addition, a review may increase public confidence in assessment findings. In particular, the adequacy of the environmental information collected and presented is checked. A review is needed also, because the proponent in whose interest it is to obtain permission for a certain initiative or its alternative cannot be expected to view initiatives completely dispassionately (Wathern 1988). In Table 6, the items that have been included in the literature as among best practice in scoping are presented.

Slootweg et al. (2006) also argue that, in the case of EIA, the reviewers should be independent and different from the persons/organisations preparing the environmental impact statement.

Table 6: Best practice for review (compiled from the work of Bagri et al. 1998; Slootweg et al. 2006; and Rajvanshi 2010)

-
- checking and ensuring that biodiversity issues are adequately addressed
 - ensuring that the information provided is sufficient for the purpose of decision-making for which it is prepared
 - ensuring that the design of the initiative complies with the relevant standards and policies, or, where official standards do not exist, with standards of good practice
 - evaluating whether the impacts are acceptable from a biodiversity standpoint
 - including stakeholders’ / affected groups’ involvement
 - ensuring that concerns and comments of stakeholders / affected groups are adequately considered and included in reporting
-

Monitoring

The planning of monitoring precedes implementation of the initiative, and the implementation of monitoring follows the decision on implementation (Bagri et al. 1998). Monitoring is particularly important in view of the uncertainty surrounding many elements of biodiversity (Southerland 1995). In SEA, monitoring is

used to address uncertainty (Partidário 1999; Morrison-Saunders and Arts 2004). The monitoring programme prepared during impact assessment should outline the monitoring needs and possibilities. Table 7 presents items cited in the literature as best practice in scoping.

Table 7: Best practices for monitoring (compiled from works by Morris 1995; Bagri et al. 1998; Treweek 1999; Treweek et al. 2005; Morrison-Saunders and Arts 2004; Sloomweg et al. 2006; Lee 2006; Fischer 2007; and Rajvanshi 2010)

Treweek et al. (2005) argue that monitoring frameworks should identify what biodiversity information is needed for monitoring, what indicators/measures are used, how much information should be collected and by whom,

thresholds for triggering remedial action, and mechanisms for disseminating the biodiversity information collected (for, to take an example, a second generation of proposals). This framework can be a monitoring plan for an individual initiative or an existing framework for monitoring as part of another planning or monitoring mechanism. Monitoring activities are among the least developed elements in many assessment systems, despite being central to their long-term overall effectiveness (Morrison-Saunders et al. 2003; Lee 2006). Furthermore, the active involvement and participation of (in particular, local) communities still tends to be rather limited in the monitoring of biophysical issues (Morrison-Saunders and Arts 2005).

Table 7: Best practices for monitoring (compiled from works by Morris 1995; Bagri et al. 1998; Treweek 1999; Treweek et al. 2005; Morrison-Saunders and Arts 2004; Sloomweg et al. 2006; Lee 2006; Fischer 2007; and Rajvanshi 2010)

-
- focus on those elements of biodiversity most likely to change as a result of the initiative
 - provision of feedback as to whether the approved initiative and its accompanying mitigating and enhancing measures have been satisfactorily implemented
 - evaluation of the predictions' validity – that is, whether the type and level of impacts – positive and negative – predicted have occurred (and whether any unexpected significant impacts have occurred) and thus managing the uncertainty related to the predictions
 - provision of early warning of unpredicted impacts
 - establishment of cause–effect relationships and indication of the relationship between the baseline and the affected biodiversity and ecosystem services
 - use of indicators that are specific, measurable, practicable, relevant, and timely
 - provision of information for periodic review or alteration of the initiative
 - ensuring that the initiative agreed upon meets with the set objectives, regulatory conditions, standards, and conditions concerning biodiversity
 - provision of information for local people (users of ecosystem services)
 - *ex post* evaluation reviewing the effectiveness and performance of the environmental assessment process
 - auditing of the impact prediction process by assessing accuracy of predictions
 - recommendations concerning where either the implementation of the initiative needs to be strengthened or it may need to be amended
 - provision of feedback for the design of new initiatives
 - development of a scientific basis for ecological impact assessment by, for example, enhancing understanding of variation
-

3.4 Actors and their roles in ecological impact assessment

In ecological impact assessment, the following actor groups can be identified: the proponent or planner preparing an initiative, a consultant

(with or without sub-consultants) and external experts from institutions such as universities, a reviewing authority if part of the legislative framework, and stakeholders. The core actors are those directly involved in the assessment process. Stakeholders include local direct users

of ecosystem services (e.g., farmers, foresters, and fishermen) or direct local beneficiaries of the initiative who gain economic and social benefits or enhanced ecosystem services because of an initiative (e.g., users of water from a municipal water plant or a recreation route), indirect local beneficiaries (e.g., residents protected against flooding), and beneficiaries who are distant in space and time (e.g., people receiving a food supply and carbon sequestration at regional, national, or global level). There are also so-called absent stakeholders – present in future – and

general stakeholders, such as the general public, linked to transparency of the assessment process, along with non-governmental organisations (NGOs) and government authorities responsible for conservation of biodiversity, formal and informal institutions representing affected people, and scientific institutes (Slootweg 2005; Slootweg et al. 2006).

The actors and their roles are categorised in Table 8. Core actors in EIA and SEA act in both the private and the public sector, as do stakeholders.

Table 8: Actors and their roles (compiled from works by Slootweg 2005; Slootweg et al. 2006; and Lawrence 2007c)

	Public sector	Private sector
Core actors of ecological impact assessment	<p><i>'Public-sector insiders</i></p> <ul style="list-style-type: none"> • Planning authorities, from local to national level <ul style="list-style-type: none"> – preparation of the plan, programme, or policy – interest in approval • EIA or SEA reviewer (if legislated in a national environmental assessment system) <ul style="list-style-type: none"> – reviewing environmental statements (EISs) of EIA and environmental reports of SEA, and evaluating the adequacy of the process • Permitting and approving competent authorities <ul style="list-style-type: none"> – granting permits for projects, and approving plans, programmes, or policies – evaluating the final significance of impacts 	<p><i>'Private-sector insiders</i></p> <ul style="list-style-type: none"> • Project proponent <ul style="list-style-type: none"> – preparation of a project – interest in a permit • Consultants (main and sub-consultants) <ul style="list-style-type: none"> – methodological choices – use and production of technical information – baseline studies and impact prediction – documentation • External experts <ul style="list-style-type: none"> – providing scientific information on special biodiversity elements • Independent review panels (if legislated for in the national environmental assessment system) <ul style="list-style-type: none"> – reviewing environmental statements (EISs) of EIA, and environmental reports of SEA
Stakeholders in ecological impact assessment	<p><i>'General public stakeholders in biodiversity issues</i></p> <ul style="list-style-type: none"> • Ministries and government agencies <ul style="list-style-type: none"> – responsibility for maintenance of biodiversity – preparation and interpretation of legislation and regulatory standards – production of data • Environmental authorities from local to national level <ul style="list-style-type: none"> – guidance of the substantive and procedural content of ecological impact assessment • Universities and research institutes <ul style="list-style-type: none"> – production of information on the scientific basis for biodiversity and ecosystem services 	<p><i>'Users and beneficiaries of ecosystem services</i></p> <ul style="list-style-type: none"> • Users of provisioning ecosystem services (farmers, foresters, fishermen, and companies) <ul style="list-style-type: none"> – direct use of services • Beneficiaries of regulating and cultural ecosystem services (inhabitants and the public) <ul style="list-style-type: none"> – expressing public issues and preferences <p><i>'General private stakeholders in biodiversity issues</i></p> <ul style="list-style-type: none"> • Interest groups and associations (environmental NGOs and business associations) <ul style="list-style-type: none"> – conservation and user interests • Distant users and beneficiaries <ul style="list-style-type: none"> – need and use for ecosystem services on other spatial scales and in future; effect on sustainable choices

Valuation of biodiversity and ecosystem services can be completed only in negotiation with stakeholders. The experts can model biophysical changes, and local stakeholders can evaluate the effects. However, the evaluation is dependent on the planning situation in question and is, in part, already set beforehand. Bagri et al. (1998) and Slootweg (2005) see that forms of valuation are expressed by legal procedures, official and social norms, and policy objectives. The norms might have to do, for example, with noise levels or water quality. In many cases, legal norms do not exist or do not have a specific biodiversity and ecosystem services perspective. Social norms are more complex, because they depend on specific actor groups and their values and beliefs surrounding biodiversity.

Actors may understand biodiversity in several, different ways, depending on their values, their actor group, and their educational background. Wegner et al. (2005) found in an Australian study interviewing 19 EIA practitioners (including proponents and consultants, EIA review officers, and representatives of government departments and environment-focused NGOs) that half of the actors applied the traditional definition of the 1960s and 1970s, with biodiversity focusing on species and their abundance, richness, and diversity plus genetic and ecosystem diversity. The other half used a comprehensive definition including also landscape-level diversity, evolutionary processes, spatial and temporal considerations, and consideration of cumulative impacts. All NGO representatives held the traditional view, and all of the review officers held the comprehensive one. Only a small number of consultants used a comprehensive concept of biodiversity. These concepts were found to be reflected in practical approaches and views concerning what information should be included in ecological impact assessment (Wegner et al. 2005). If only half of the actors regard specific circumstances (e.g., spatial and temporal considerations) and cumulative effects as important, differences in views lead to misunderstandings and differing expectations as to what is considered significant from the beginning (scoping and deciding what biodiversity element to study) to the end of the

process (evaluating the significance of adverse impacts and their monitoring needs).

3.5 Ecological impact assessment tools

Environmental impact assessment methodology is the overall strategy used to manage an impact assessment, together with methods and techniques used to examine specific issues within the impact assessment. The methods are approaches for tackling more specific issues, and the techniques are the technical tools used within methods to achieve certain ends (Morgan 1988). However, the term ‘tool’ is used in many different fields of science and practice. As a result, the concept of tools is ambiguous and may be accorded a wide variety of meanings – for example, as technical and scientific equipment and methods for gathering, processing, storing, or displaying information. In some systems, process tools, such as EIA and SEA themselves, are regulated with respect to process and content to a degree that makes them close to scientific methods. In other cases, they could better be termed ‘approaches’. There exist a wide array of approaches and tools related to sustainability in environmental assessment (Sheate 2010). Furthermore, ‘toolboxes’ for planning and management have been developed to illustrate the breadth of the term. The concept of tools might also be used to denote ‘process packages’, which may contain a variety of processes, analyses, and methods (Emmelin 2006).

A large number of methods and tools are available for use in environmental assessment. Already in the 1980s, 350 methods and tools had been identified (Lee 2006). A wide range of technical tools can be used to identify and analyse effects on biodiversity, among them simple qualitative checklists, matrices and flowcharts, questionnaires, expert opinions, descriptive cartographic overlays and simulations, partly or totally quantitative GIS analyses, and complex quantitative models (Treweek 1999). However, in ecological impact assessment, hardly any tools are used (de Jongh et al. 2004; Gontier et al. 2006). In SEA, the range of methods used is actually very limited (Therivel and

Wood 2005; Fischer 2007). In addition, most efforts to develop aids for ecological impact assessment have been somewhat supply-driven and lessons learnt in the use of tools and their feasibility in real life through example cases are scarce (Pritchard 2005).

Methods and tools are highly dependent on the assessment tasks at hand. They are also embedded in the theories, assumptions, and planning cultures on which they are based (Emmelin 2006; Bjarnadóttir 2008). Some tools may have a background in natural sciences and technology or work within scientific or administrative frameworks where assumptions of highly rationalist decision-making models do not come into contact with research on planning, decision-making, or implementation (Emmelin 2006). Therivel and Wood (2005), Fischer (2007), Emmelin (2006), and Bjarnadóttir (2008) list a wide array of environmental assessment tools and their uses. Because the choices between methods and tools may have a major influence on the quality of the overall assessment, their selection needs to be made in a systematic manner and already in the scoping phase (Lee 2006). Certain characteristics of good environmental assessment tools can be identified. Good tools should

- be able to be implemented rapidly,
- help to improve the planned action,
- focus on key impacts,
- cope with uncertainty,
- take account of indirect and cumulative impacts,
- suggest and compare alternatives, and
- be robust and easily understandable (Therivel and Wood 2005).

Factors to take into account when choosing from among alternative tools may include

- the type of the assessment task,
- the level of detail and degree of accuracy to which the task needs to be performed,
- the consistency of each method selected with the other assessment methods to be included within the methodology,
- the data, expertise, time, and other resource requirements of each method, and

- the transparency, intelligibility, and credibility of each method as perceived by the decision-makers and other stakeholders (Lee 2006).

Different methods and tools can be used in different phases of the assessment. Morgan (1988) and Fischer (2007) recommend the following broad tool types:

- screening: indicators, checklists, threshold lists, expert judgements/opinions, communication/reporting, and preliminary studies
- scoping: indicators, checklists, matrices, surveys, participation, communication, consultations, expert opinions, and SWOT analysis (examining strengths, weaknesses, opportunities and threats)
- impact assessment/reporting: indicators, various types of checklists (descriptive, questionnaire, etc.), matrices, surveys, communication, participation, consultation, network and flow diagrams, statistical analyses, overlay maps, forecasting, expert opinions, and SWOT analysis
- review: indicators, consultation, participation, and expert opinions
- monitoring: indicators, surveys, communication/reporting, and expert opinions.

Geneletti (2004) presents quantitative methods by using spatial indicators to predict and quantify direct ecosystem loss and fragmentation in ecological impact assessment for roads. Atkinson (1985) presents a range of habitat-based quantitative methods measuring baseline conditions and quantification of predicted impacts. According to a literature review concerning GIS-based ecological models that was carried out by Gontier et al. (2006), there exist models with potential for impact prediction on landscape and regional levels especially with respect with fragmentation. They argue that the models could provide a quantitative approach and allow impact predictions to be prepared not for the study area itself so much as also for the surrounding environment. However, there are many requirements and limitations of quantitative models and methods; one might consider, for instance, the availability of GIS and other

data, expert knowledge, understanding of the methods and their limitations, the level of detail required in the assessment, and resources (Atkinson 1985; Gontier et al. 2006).

There is a clear gap between ecological research on models and their use in practice. Generally, the complex models are not used in real-life ecological impact assessments. Rare exceptions of more than very simple tool applications are found in research-oriented EIA and land-use SEA plan case studies, where a case study has been carried out in detail as part of wider research work or tool development (e.g., Fernandes 2000; Geneletti 2002; Malarach and Marul 2006; Mörtberg et al. 2007). One biodiversity- and ecosystem-services-related tool's development and use is described by Cooper (2010), presenting experiences of network analysis based on the use of network diagrams, which demonstrates the ecosystem services provided in the baseline situation of the local and regional green infrastructures and how the ecosystem services would change in certain management scenarios (Cooper 2010). Preliminary network diagrams can be used in stakeholder workshops where participants are able to provide feedback and ideas or modify the diagrams. However, developing network diagrams requires an ecosystem services typology, precise land-cover information, and understanding of the relationships between variables of land-use or land-cover categories and the ecosystem services provided. Among the shortcomings of network analysis are that there is a limit to the amount of information that can be shown in a complicated network diagram if one wishes to keep it understandable and that quantification and a spatial dimension are absent.

4 The Finnish legal and procedural framework for ecological impact assessment

The ecological impact assessment regime in Finland, as a member state of the EU since 1995, follows the European Union legislation consisting of the EIA Directive (CEC 1985)

as amended in 1997, 2003, and 2009 (CEC 1997, 2003, 2009); the Habitats Directive (CEC 1992); the directive on the conservation of wild birds, referred to also as the Birds Directive (CEC 1979); and the SEA Directive (CEC 2001). The EIA Directive (CEC 1985) has been transposed to Finnish legislation through the Act on Environmental Impact Assessment Procedure (1994), referred to later herein as the EIA Act, which entered into force in September 1994. The Decree on Environmental Impact Assessment Procedure (1994, with amendments in 1995), referred to hereinafter as the EIA Decree, complemented the EIA Act as the other main component of Finnish EIA legislation. The EIA Act was revised in 1999 (Act on Environmental Impact Assessment Procedure 1999), and at the same time the EIA Decree was renewed (Decree on Environmental Impact Assessment Procedure 1999). A second revision of the EIA Act and renewal of the EIA Decree were completed in 2006 (Act on Environmental Impact Assessment Procedure 2006; Decree on Environmental Impact Assessment Procedure 2006). The revisions of the EIA Act were based on the amendments to the EIA Directive (CEC 2003). The assessments required by Article 6(3) and Article 6(4) of the Habitats Directive (CEC 1992) have been transposed to Finnish legislation through the Nature Conservation Act (1996). This whole process, including screening and statements, is referred to below as the Natura 2000 assessment, and the phase including the scoping and actual assessment is referred to as the appropriate assessment (AA). The SEA Directive (CEC 2001) was transposed to Finnish legislation by the Act on the Assessment of the Impacts of the Authorities' Plans, Programmes and Policies on the Environment (2005), later called also the SEA Act and Decree on the Assessment of the Impacts of the Authorities' Plans and Programmes on the Environment (2005), which addressed all plans, programmes, and policies but not land-use plans. This is referred to below as the SEA Decree. The requirements of the SEA Directive (CEC 2001) not already incorporated into the Finnish land-use legislation, which was prepared in parallel to the SEA Directive (CEC

2001), were transposed through changes to the Land Use and Building Act (1999, amended in 2005) and Land Use and Building Decree (1999, amended in 2005). In addition, other sectors' legislation affects the content of ecological impact assessment (see *Article IV* of this work).

The assessment procedures of Finnish EIA, Natura 2000 assessment, and SEA in land-use planning follow, by and large, the procedural phases presented above in chapter 3.3.2, but differences exist between the procedures in the extent of their phases, the documents required, and the role of authorities. All directives of the EU regarding ecological impact assessment leave broad choice for the form and content of the assessment process. The number of EIA procedures varies, being 30–50 a year. Statistics submitted by regional environment centres on appropriate assessments indicate that in 2001–2005, 10 assessments were carried out per year but the figure can be larger (see *Article III* of this work). The number of assessments requiring SEA under the SEA Act (2005) totals 10–20 per year; the corresponding figure for the Land Use and Building Act (1999) is 1,500 assessments per year, of which 100 are for local master plans.

4.1 Finnish EIA procedure

In Finnish EIA, the assessment procedure is always applied to major projects and their alterations as listed in the EIA Decree (2006), without any screening, and also is followed on a case-by-case basis for projects, according to the screening criteria listed in the above-mentioned decree. Among the latter are the biodiversity-linked screening criteria in the EIA Directive (CEC 1985), mentioning elements such as the existing land use; the abundance, quality, and regenerative capacity of natural resources in the area; and the absorption capacity of the natural environment, especially in certain ecosystems, on protected nature sites, and in densely populated areas. The definition of environmental impacts covers direct and indirect impacts of the project on several factors, including fauna, flora, and biological diversity. In addition, the Nature Conservation Act (1996) regulates indi-

rectly which elements of biodiversity shall be taken into account in the assessment.

The developer provides adequate information for the screening, and the decision on the application is made by an EIA authority. Until 2010, there were 13 regional environment centres as EIA authorities. Since the beginning of 2010, the regional governmental administration has been organised into 15 new regional centres for economic development, transport, and the environment, also called ELY Centres. The EIA authority is a liaison authority (a role differentiated from other duties of the centres), supervising the enforcement of the EIA Act (2006), co-ordinating the EIA process, and being responsible for the quality control of the process but not for decision-making (*inter alia*, making of permit decisions, which is the task of the competent authorities). This use of a designated EIA authority, which is not a competent authority in decision-making, distinguishes the Finnish EIA system from the systems in other jurisdictions (Pölonen et al. 2011). The strong role of an EIA authority has been confirmed by Finnish court rulings, where significant weight has been given to its statement in reviews of the adequacy of the assessment reports (ARs). If the EIA authority considers the environmental impact assessment studies in a case adequate, it is highly unlikely that the court will reverse the decision for reasons of poor EIA quality (Pölonen et al. 2011). Since the 2006 amendment of the EIA Act (2006), it has been possible to complain about a decision concerning a project on the basis of the inadequacy of the EIA; before that, legal complaints were possible only in the event of assessment being absent.

The EIA authorities review scoping documents, in Finland called assessment programmes (APs), and environmental impact statements, in Finland called assessment reports (ARs), within their region. The assessment process begins when a developer delivers an AP to an EIA authority. The authority ensures that the necessary statements on the AP are requested and provides opportunities for expressing opinions on the AP. The EIA authority issues its statement (or 'official opinion') on the AP, including a summary of other statements and

opinions to the developer. If necessary, the EIA authority points out in the statement the issues with the AP that must be resolved. After receiving the statement, the developer carries out the assessment and prepares an AR. Hearings, collection of opinions, and statements co-ordinated by the EIA authority follow the publication of the AR. Finally, the EIA authority makes its own statement on the AR and its adequacy. This ends the official EIA process. The requirements set under the EIA Decree (1999)'s sections 9 and 10 are listed in detail in *Article I* of this work. The EIA Decree from 2006 added information on the baseline to the explicit requirements for the AP, and for the AR it added comparison of alternatives, description of the phases of the procedure (including the participation), and description of how the statement of the EIA authority on the AP has been taken into account, which were missing earlier. In practice, in addition to what the EIA legislation requires, there is typically considerable informal communication between the developer, the EIA authority, and the public (Pölonen et al. 2011).

After the EIA, a permit decision or comparable decision on a project has to include information on how the AR and the statement on it have been taken into account. Since the decision-making is not a part of the EIA, the Finnish system has been criticised for poor linkage between EIA and decision-making (Pölonen 2006; Pölonen et al. 2011). Under EIA legislation, a competent authority is not obliged to follow the recommendations of the AR or required to minimise the project's negative effects on the environment. The EIA Directive (CEC 1985, Article 8) and EIA Act (1994, Section 13.2) require that information gathered in the EIA process be taken into account in the permit procedure. However, this is only a procedural requirement and does not determine how they are to be taken into account in decision-making; therefore, they do not in themselves strengthen the ecological controls (Pölonen et al. 2011). Pölonen (2006) and Pölonen et al. (2011) do not see this procedural nature of EIA as a problem in cases where EIA is connected to decision-making. The majority of the activities subject to EIA also require an integrated environmen-

tal permit under the Environmental Protection Act (2000), ensuring that significant effects on the environment are prevented. However, the Environmental Protection Act (2000) mainly concerns itself with emission discharge into the air and with water and soil pollution so may omit the biophysical changes from the permit consideration (Pölonen et al. 2011). This discrepancy pertains to effects on biodiversity and ecosystem services. Furthermore, for activities not subject to environmental permit procedure but requiring permit procedures that involve only limited consideration of the environment, such as permission for running power lines, environmental impacts can be disregarded. To raise the level of substantive requirements for EIA, Pölonen (2006) and Pölonen et al. (2011) proposed amending the EIA Directive (CEC 2009) and, respectively, the Finnish EIA Act (2006) via preconditions similar to those of the Habitats Directive (CEC 1992, Article 6 (3) and the Finnish Nature Conservation Act (1996, Section 66) such that in cases of significant adverse environmental impact, the permit would be denied or the project carried out only in the case of lack of alternative solutions and for reasons of overriding public interest. The typical Finnish EIA process is presented in Figure 4.

4.2 Finnish Natura 2000 assessment procedure

The present Finnish Natura 2000 network consists of 1,857 sites (Ministry of the Environment 2011). According to the Finnish Nature Conservation Act (1996), the duty of assessment of relevant projects and plans has been enforced since the first proposal of sites to the European Commission, in 1998.

The Habitats Directive (1992) does not explicitly regulate how the assessment procedure should be carried out. Although the European Commission has published guidelines (European Commission 2000, 2001), there are many and different, even contradictory, interpretations at the national level with respect to what the Natura 2000 assessment process should include and how it should be carried out on different

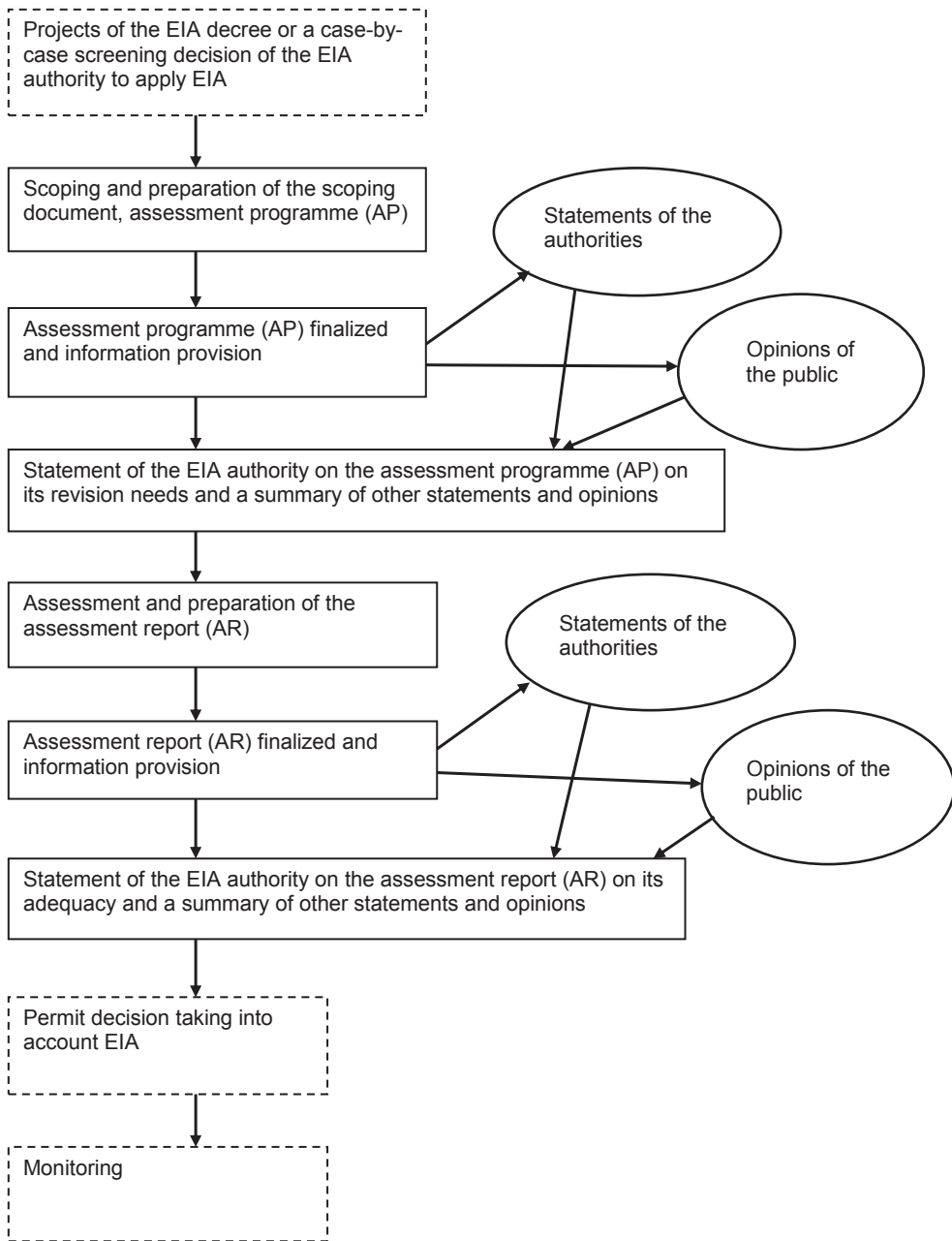


Figure 4. Typical EIA process in Finland.

planning levels (European Commission 2009a, 2009b; Therivel 2009; Peterson et al. 2010). In addition, the European Commission has recently published sector-specific guidelines for the Natura 2000 assessment process covering wind energy, non-extractive industries, and ports and estuaries, in which it emphasised the importance of proactive strategic-level spatial planning and that it is in the process of preparing guidance on inland waterway transport and aquaculture (European Commission 2011).

With respect to the content of the assessment, Article 6(3) of the Habitats Directive (1992) mentions likeliness of effects, the significance of the effects, effects on the site's conservation objectives, individual and 'in combination' effects of other projects or plans, appropriateness of the assessment, competent authorities' duty to ascertain that the activity will not have significant adverse effects on the Natura 2000 site before permitting or approving said activity, and the opinion of the general public. Article 6(4) regulates approval of activities with negative impacts on Natura 2000 sites only when there are no alternatives and at the same time there is overriding public economic or social interest and are compensatory measures to compensate for the loss of conservation values. The Finnish Nature Conservation Act (1996, sections 65 and 66) also explicitly states that the assessment concerns plans and projects both on and outside the sites. In addition, the Nature Conservation Act (1996) requires that the competent authority making decision on the plan or project ask for a statement (the 'official opinion') from the regional environmental authority (formerly the 13 regional environment centres and now the 15 ELY Centres) before making a decision. These statements function as a quality control for the assessment by expressing views on the adequacy of the appropriate assessment report and significance of the impacts. In addition, the Nature Conservation Act (1996) states that in cases of significant adverse effects, the plan or project can be accepted only after ratification at the national level by the Council of State.

The guidance of the European Commission (2001) distinguishes four stages in the assessment of plans and projects significantly affect-

ing Natura 2000 sites: 1) screening, 2) appropriate assessment, 3) assessment of alternative solutions, and 4) assessment of compensatory measures. In the Finnish Natura 2000 assessment procedure, there are two phases. The first is a screening equivalent to stage 1 in the EU guidance (European Commission 2001). The screening is carried out when the planner is unsure whether significant adverse effects on a Natura 2000 site are likely. If significant effects are very likely, the assessment proceeds straight to the appropriate assessment. If the impacts are considered in the screening not to be significant, a short report of one to three pages is written and a competent authority may consider approval of the project or plan. Screening reports concluding that likely significant adverse effects do not exist are not collected in any official statistics. Therefore, there are no statistics on how many of them are carried out annually. However, there has been a tendency in practice to carry out the whole Natura 2000 assessment as a rather broad screening exercise without having a full appropriate assessment or a statement from a regional environmental authority (Similä et al. 2010). If the screening points to possible significant adverse effects, the assessment proceeds directly to the AA stage. The appropriate assessment usually includes both stage 2 and part of stage 3 in the EU guidance. With respect to alternatives, the Finnish procedure is aimed at identifying that alternative not causing significant adverse effects and doing so during the planning so as to avoid a wholly new assessment. The content of the AA follows the guidelines prepared in the joint work of the environmental authorities with other national, regional, and municipal authorities including representatives of consultants and a nature NGO, in an attempt to follow the EU guidelines (European Commission 2000, 2001) as coherently as possible (Söderman 2003). The guidance requires assessment in terms of the conservation objectives for the Natura 2000 site on the species and habitat type level and of integrity as a whole, both individually and cumulatively (Söderman 2003). No Natura 2000 assessment process in Finland has proceeded to stage 4, including the identification of compen-

satory measures, because the planning has usually been altered or stopped upon recognition of the need for a permit from the Council of State and for compensatory measures. In one case, involving a major harbour project, the Council of State permit and the need for compensatory measures were controversial but the process led to the conclusion that the impacts were not significant (Nordberg 2007).

The Finnish system is flexible in allowing combination of the processes of EIA, SEA, and AA as long as the AA report can be clearly separated out in the assessment documents and the statements given on the AA can be separated from other statements (Söderman 2003). When the AA report is ready, the proponent or planner sends it to the competent authority, which must request a statement from the regional environmental authority. The statement can propose rejection of the project/plan or conditions for approval, such as selection of a certain alternative (if more than one were presented) or additional mitigation measures. It can also deem the whole AA report inadequate. After receiving both the AA report and the statement, the competent authority can approve or reject the project or plan. If the AA report and statement reach the same conclusion, deviation from it is not permitted. If, on the other hand, they differ, the decision-making authority can use its discretion concerning which of the two it considers to be correct (Nordberg 2001). In light of the results of Pölonen (2006, 2007) related to the decisive role of statements by the EIA authority, it could be presumed that statements of the regional environmental authorities would have the same weight on decisions made on the basis of the Natura 2000 assessment. However, this has not been studied or proved.

If the result of the consideration is that adverse effects are significant, the permit or plan is rejected, so when only one planning option is presented in the proposal, the assessment process will start again, with a new option. If the result of the consideration is that the effects are non-significant, the project or plan or can be approved or the permit admitted. The Natura 2000 assessment process differs from other assessment processes wherein assessment

findings have to be taken into account in decision-making – in Natura 2000 assessment, there is a direct precondition to decision-making: approval cannot be granted if the results show significant adverse impacts. Because of these strict requirements, Therivel (2009) argues that the Natura 2000 assessment is a decision-making rather than a decision-informing tool. It also guides decision-making strictly toward an ideal mitigation hierarchy favouring avoidance first, reduction of impacts next, and compensation as the last resort.

A typical Natura 2000 assessment process in Finland is presented in Figure 5.

4.3 Finnish local master planning SEA procedure

Because this work studies the ecological impact assessment in land-use planning as part of the SEA application, the SEA process is presented within the framework of the Land Use and Building Act (2009). The SEA process under the Finnish SEA Act (2005) is described in detail by Söderman and Kallio (2009).

The Finnish spatial planning system as defined in the Land Use and Building Act (1999) comprises three levels of spatial planning. These are hierarchical in nature: higher-level plans must be taken into account when authorities prepare plans at lower levels. The three levels of plans are 1) the regional plans prepared by the regional councils, 2) the local master plans, and 3) the local detailed plans prepared by municipalities. The substantive requirements regarding biodiversity are set in broad terms for each plan level. For example, at the regional level, the requirements are ecological sustainability of land use and protection of natural values. In addition, sufficient areas suitable for recreation should be included (Land Use and Building Act 1999, Section 28). At the local master plan level, these requirements include ecological sustainability of the community structure and the protection of natural values (Land Use and Building Act 1999, Section 39). Here too a sufficient number of areas suitable for recreation is required. In addition, other requirements connected to ecosystem services are mentioned

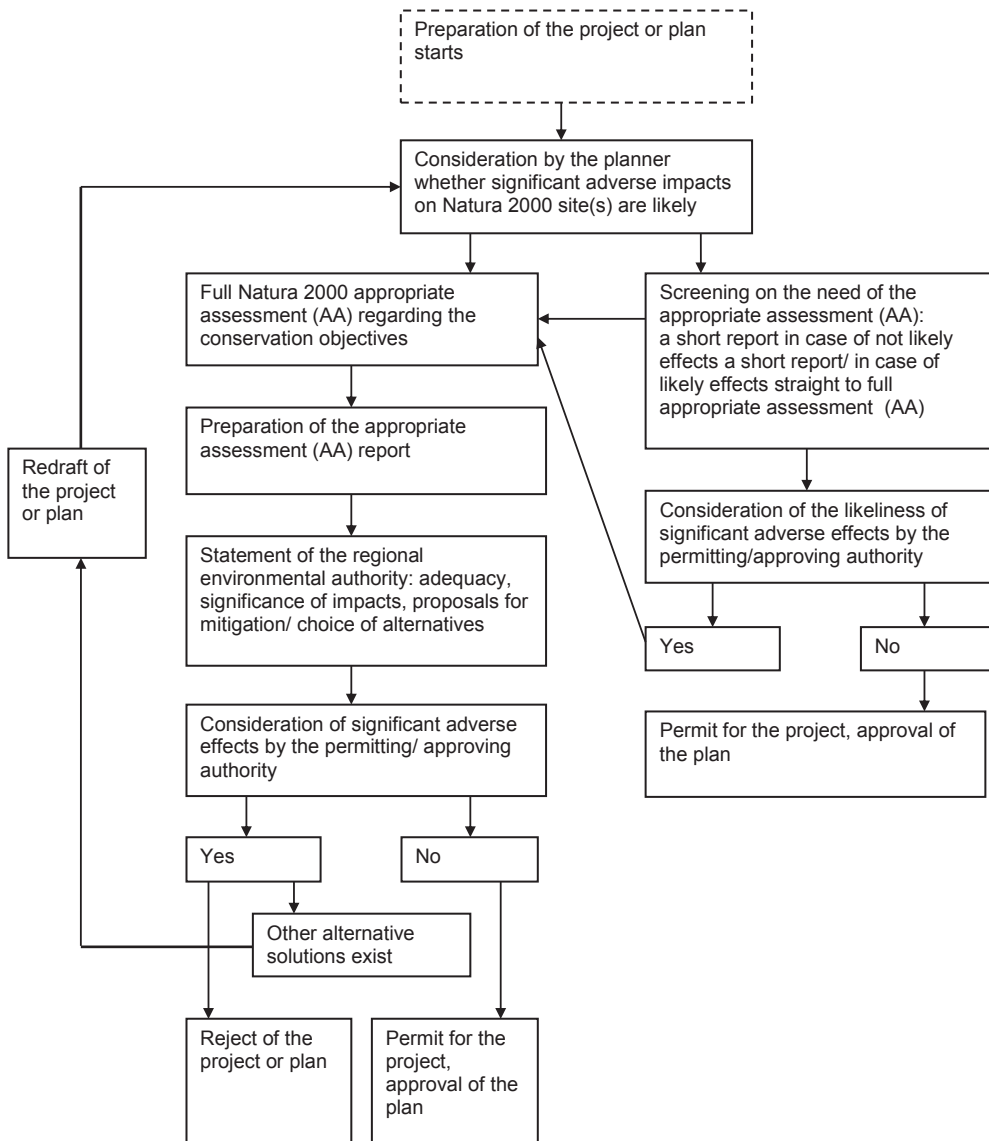


Figure 5. The typical Natura 2000 assessment process in Finland.

– e.g., landscape values and good water supply and drainage. In addition, the Council of State has set national land-use objectives, which steer preparation of plans. With respect to biodiversity, these include promotion of preservation of valuable and sensitive nature areas, ecological connections – both between individual protected areas and between protected and other valuable nature areas – and use of the network of protected areas in recreation that does not compromise the conservation objectives (Valtioneuvoston päätös... 2008).

The impact assessment procedure concerned with local land-use plans commences when preparation of the plan begins. Negotiations between authorities are set up with regional environmental authorities, the ELY Centres, and some other invited authorities. All local master plans with importance in terms of nature values must be negotiated. In this phase, information is provided to the public and stakeholders. Unlike the SEA Directive (2001) and the Finnish SEA Act (2005), the Land Use and Building Act (1999) requires preparation of a scoping report, termed a participation and assessment scheme, at the beginning of the planning process. The scheme should cover participation and interaction procedures and a plan for the assessment of the plan's impacts. The authority making the plan may negotiate with the regional environmental authority and other interested parties on the adequacy and implementation of the scheme. After the planned baseline studies are carried out, alternative plan options are addressed and their environmental impacts are assessed. In addition, exchange of information and informal negotiations may be arranged if desired. Then a plan proposal including a plan statement, which is equivalent to the environmental report of the SEA Directive, is prepared and published – made available for statements from the authorities and opinions of the public. The municipality then sends responses to the parties who objected to the plan, mediates negotiations between authorities, and finally approves the plan. If the plans cover several municipalities, the Ministry of the Environment ratifies them after approval.

Specific requirements for the impact assessment are set forth in Section 9 of the Land Use and Building Act (2009), according to which environmental impacts of the plan and its alternatives have to be assessed to the necessary extent for the area on which the plan has material impacts. Both 'necessary extent' (referring to adequacy) and 'material impacts' (referring to significance of impacts) have been explained further in the Land Use and Building Decree (1999, sections 1 and 17): the investigation must provide the data necessary for assessing the significant direct and indirect impacts of the plan's implementation on, *inter alia*, plants and animals and biodiversity and must present the issues in a manner and extent suitable in view of the purpose of the plan and interaction. The content requirements for the plan statement of a local master plan are set by the Land Use and Building Decree (1999, Section 17). These include

- 1) an account of circumstances in (i.e., description of) the area, its environmental features, and changes in them, and of other information on the area subject to planning that is essential for investigation and assessment of the plan's impact,
- 2) the starting point for the planning, the aims of the planning, and proposed options,
- 3) a summary of the investigations carried out to assess the plan's impact,
- 4) the plan's impact on community structure, the built environment, nature, the landscape, traffic arrangements (especially public transport) and technical services, the economy, health, social circumstances, and culture, and any other significant impacts;
- 5) an account of the plan's relationship to national land-use objectives, the regional plan, the current local master plan, and the local authority's other planning,
- 6) the stages of planning, including participation and interaction procedures, and a summary of the comments expressed in the various stages of the planning process,
- 7) the key content and principles of the selected planning option and an account of how the results of impact assessment and the comments expressed have been taken into

account and of account mitigation measures to prevent potential negative impacts of the plan,

- 8) the schedule for and monitoring of the plan's implementation, and
- 9) when needed, schemes steering plan implementation.

In addition, Section 17 of the Land Use and Building Act (1999) states that a summary of the information provided in the above categories should be included.

The typical planning procedure for local master planning in Finland is presented in Figure 2 in *Article IV* of this work. The phases and their content vary, depending on the extent of the plan and on the municipalities' planning practices.

5 Material and methods

The empirical analyses utilised four different sets of survey and interview data in examining ecological impact assessment in three distinct planning and assessment processes, including EIA, Natura 2000 appropriate assessment, and municipal local master planning SEA. Components of the assessment were analysed, and the results are reported upon in *articles I, II, III, and IV* of this work. The first three articles listed present reviews of ecological impact assessments carried out via document analysis and a case study. The fourth study (see *Article IV*) gathers data on actor views on ecological impact assessment practices. Table 9 presents an overview of data from *articles I–IV*. In the fifth study (see *Article V*), a conceptual approach and practical tool for improving the knowledge foundation and practices of ecological impact assessment by using GIS-based information and map presentations was developed.

5.1 Review of environmental assessment reports

A qualitative document analysis of environmental assessment reports prepared as a result of EIA procedure was carried out to examine ecological impact assessment practices. All

told, 38 reports were selected from the archives of the Finnish Environment Institute. The sample was selected to represent the project types that cause the most severe impacts on biodiversity, such as road and railway projects, electricity transmission lines, peat production, harbours and shipping channels, peat extraction, hydroelectric power plants, and water intake and flood control. The oldest reports were from 1995, when the EIA was a relatively new practice in Finland, and the most recent is from 2001, when EIA had been practised for almost eight years.

The reports were analysed through the use of 43 review questions to examine how biodiversity issues were described in obligatory environmental reporting related to the requirements of the Finnish EIA legislation in effect at the time (EIA Act 1994, 1999; EIA Decree 1994, 1999) and the concepts and best practice drawn from the ecological impact assessment literature. The review questions were divided across seven categories, characterising the treatment of the project proposal, baseline of ecological issues, impact prediction, mitigation and monitoring, cumulative impacts, and map presentations. Each report was reviewed by means of the review method outlined by Atkinson et al. (2000), systematically examining different parts of the reports and then categorising the information provided in the report into three levels of information provision. At the first level, 'satisfactory', most decision-makers would understand the implications of the project with regard to the issue addressed. At the second level, 'partly satisfactory', the issue was addressed to some extent but most decision-makers would probably remain unsure of the project's implications. At the third level, 'not addressed', the issue was not mentioned at all. An ecological and biodiversity index (EBI) was used to quantify the qualitative answers to the review questions. The index was calculated on the basis of the number of review questions that received 'satisfactory' or 'partly satisfactory' answers.

This study method has some limitations, firstly associated with the document analysis, which reveals only those issues reported on in the ecological assessment process, and it is

Table 9: Overview of data

Data:	Review of environmental assessment reports	Case study of a large-scale environmental impact assessment process	Comparative review of Natura 2000 appropriate assessment reports and statements given on them	Expert interviews on ecological impact assessment in land-use planning
Data years:	1995–2001	2002–2004	1997–2005	2007
N:	38	1	73 (plus 70 statements)	20
Sample:	Targeted sample of projects with the most severe impacts on biodiversity	Ongoing EIA process for a project type with the most severe impacts on biodiversity	All appropriate assessment reports and statements recorded during the data period	Selected interviewees presenting key actors in ecological impact assessment
Coverage:	National	Administrative region of the Uusimaa environment centre	National, except for the administrative region of the South-west Finland environment centre	The regions of the five southernmost environment centres: Uusimaa, South-west Finland, Häme, Pirkanmaa, and South-east Finland
Number of projects and plan types or actor types:	<ul style="list-style-type: none"> – 13 road projects – 7 power transmission line projects – 5 peat production projects – 5 harbours and shipping channel projects – 3 water intake and flood control projects – 2 railway projects – 1 land reorganisation project 	<ul style="list-style-type: none"> – 1 power transmission line project 	<ul style="list-style-type: none"> – 1 regional plan – 20 local master plans – 6 local detailed plans – 11 road projects – 5 mining projects – 4 water intake projects – 3 peat production projects – 3 nature management and recreation projects – 2 power transmission line projects – 1 gas pipeline project – 1 land reorganisation project – 1 railroad project – 1 harbour project – 1 case of forest logging – 13 other construction activities 	<ul style="list-style-type: none"> – 10 authorities – 5 municipal land-use planners – 5 consultants
Themes:	<p>Biodiversity considerations in description of:</p> <ul style="list-style-type: none"> – project proposal – baseline – impacts – alternatives – mitigation and monitoring – cumulative impacts <p>Use of GIS/maps</p>	<p>Biodiversity considerations in:</p> <ul style="list-style-type: none"> – initial design and the choice of alternatives – screening – scoping – baseline studies – impact prediction – mitigation – monitoring – reporting <p>Evaluation of adequacy, roles of reporting, and actors</p>	<p>Biodiversity considerations in description of:</p> <ul style="list-style-type: none"> – screening – project or plan proposal – baseline – impacts – cumulative impacts – alternatives – mitigation and monitoring <p>Use of GIS/maps Evaluation of adequacy and impact significance, and roles of actors</p>	<p>Biodiversity considerations in:</p> <ul style="list-style-type: none"> – scoping – baseline studies – impact prediction – monitoring – collaboration and roles of actors
Analyses:	Qualitative and semi-quantitative	Qualitative	Qualitative and semi-quantitative	Qualitative
Reported in article	I	II	III	IV

dependent on the level of reporting. However, this was considered acceptable because the assessment report should present all relevant information without additional reports. Secondly, the review questions are subjective and inevitably reflect the reviewer's understanding of what constitutes satisfactory treatment of an issue. In addition, some review questions are interdependent, which pushes indices toward lower scores. However, all of the reports were reviewed in the same way and both the qualitative evaluation of the issues addressed and the indices calculated are comparable between reports over time. More detailed description of the material and methods, including details of the review questions, is provided in *Article I*.

5.2 Case study of a large-scale environmental impact assessment process

The case study reviewed the EIA procedure of a single large-scale linear project to find in greater depth the underlying reasons for shortcomings in ecological impact assessment practice that were detected in the previous study (*Article I*). The detailed process examined the EIA process for a 400 kV power transmission line between Loviisa and Hikiä, in Southern Finland. This project was chosen for the case study because one of the key causes of the loss of biodiversity is habitat change (MA 2005), viz. habitat loss and fragmentation commonly associated with linear projects (Byron 2000). The process was studied from 2002–2003 to May 2004 – i.e., from the scoping phase to the formal end of the assessment procedure, when the regional EIA authority gave its statement on the AR.

The material for the analysis was derived through attendance and follow-up of interest-group meetings during the EIA process; attendance of public hearings of the EIA and reading of the associated memos; and interviews with the proponent's representatives, the main EIA consultant and the sub-consultant hired for the ecological impact assessment, and the regional EIA authority. In addition, document analysis was performed for the main EIA reports and

supporting ecological impact assessment reporting.

The process was analysed through the use of 50 review questions developed on the basis of Hook and Fuller's (2002) EIA process review criteria, which ensures that the ecological assessment is reviewed from all angles, including aspects of biodiversity and stakeholder involvement. The review questions were grouped into eight categories, dealing with the ecological/biodiversity content of the initial project design, integration of the biodiversity considerations into the project alternatives, screening, scoping, ecological baseline studies, impact prediction and assessment, mitigation, follow-up and monitoring, and reporting. The analysis was qualitative. Evaluation of the treatment of biodiversity issues was based on best practice as outlined in international environmental impact assessment literature and in biodiversity impact assessment guidance (Byron 2000; CBD 2002), on the requirements of the EIA Directive (CEC 1985, 1997), and on the Finnish EIA legislation valid at the time (EIA Act 1994, 1999; EIA Decree 1999). Limitations of the method are linked to the timing of the evaluation, which reflects the practical understanding of biodiversity at the time. For example, although the ecosystem functions that represent values for society were mentioned in CBD COP 6 Decision VI/7 (2002) and listed in its Annex 3, the application of the ecosystem services concept was still rather underdeveloped in environmental impact assessment practice and, therefore, it was not explicitly included in the list of review questions. Details on the material and methods, including a list of the review questions, are presented in *Article II*.

5.3 Review of Natura 2000 appropriate assessment reports and statements made on them

A comparative document review including AAs from two different periods was carried out to assess whether problems identified in two earlier studies (see *articles I and II*) characterise also the type of assessment concentrating merely on ecological impacts rather than taking these

as only one category for impact assessment in the larger EIA process. In addition, the study focused on change over time in impact practices and content and in the impact of the statements given on AA reports. In total, 73 AA reports and 70 statements on them by regional authorities were reviewed. The first time period covered the AA reports and statements from the beginning of obligatory Natura 2000 assessment, 1997 to mid-2001. The second period covered the AA reports and statements from mid-2001 to 2005.

The reports were analysed by means of 45 review questions, addressing how biodiversity issues were described in AA reports related to 1) the Finnish Nature Conservation Act (1996) and the Habitats Directive (CEC 1992) and 2) the EU-level guidance given for appropriate assessments (European Commission 2000, 2001). The review questions were divided into eight groups, characterising the description of screening, the plan or project, the Natura 2000 site, conservation objectives for the site, cumulative effects, alternative solutions, mitigation and monitoring, and map presentations. In addition, the work identified 21 questions as those whose answers are critical for enabling determination of the significance of the impacts. The rest of the questions were regarded as involving best-practice issues of assessment. Each report was reviewed systematically in a set of steps that included review of the authority's statement on the adequacy of the reporting and significance of the impacts of the relevant plan or project on the integrity of the Natura 2000 site. The level of reporting was classed into one of three levels of information provision via the method applied in *Article I* of this work. Similarly, the Natura indices for both the essential questions (NI21) and all questions (NI45) were used to quantify the qualitative answers to the review questions.

The Natura indices of the reports were compared with the statements of the authorities on the adequacy of the reports. In addition, the views expressed on the significance of the impacts for individual projects and plans were subjected to comparison between the consultants preparing the report and the authorities issuing the statements on them. The quality in-

indices were also used for comparing assessment report quality between projects and land-use plans and between the time periods examined. Detailed description of the material and methods, including a list of the review questions, is presented in *Article III*. The work for *Article III* is a combination of two studies reported upon also in separate publications (Söderman 2001, 2007).

5.4 Expert interviews addressing ecological impact assessment for land-use planning

The roles and views of core actors involved in ecological impact assessment were examined through interviews of authorities at regional environmental centres, land-use planners with municipalities, and ecologists in companies carrying out ecological studies. Local master planning was chosen as the process for study to explore the potential of SEA; it was considered to offer more potential for inclusion of biodiversity and ecosystem services than EIA in ecological impact assessment. A further reason for this choice was that biodiversity was considered to be more broadly affected by spatial plans than by other types of planning, because spatial plans determine the extent and distribution of different land-use allocations that directly, indirectly, and cumulatively affect biodiversity and ecosystem services on different scales. In addition, local master planning, while strategic enough, is a relatively low level of decision-making, one at which it is feasible to demonstrate linkage between strategic decisions and their impacts on biophysical changes leading to changes in ecosystem services.

The material for the analysis was obtained from semi-structured expert interviews, for which experts were selected through the snowball sampling method, in which key informants are interviewed first and suggest further interviewees. Heads of the land-use departments of the regional environment centres (and more recently the ELY Centres) were used as key informants. The total number of interviewees was 20. The interviewees – 10 representatives of authorities, five planners, and five consultants

– were from towns in Southern Finland, where the development pressure is greater than in other parts of Finland and the most SEA processes are executed. The interviews were recorded and transcribed, and the answers to the questions were grouped under six research themes – to do with practices and views of the interviewees related to 1) scoping; 2) baseline studies; 3) impact prediction and use of ecological studies; and 4) consideration of biodiversity in planning, 5) monitoring, and 6) collaboration. The research themes and interview questions were derived from Finnish and international best-practice principles and criteria for ecological impact assessment, SEA, and land-use planning and from the Land Use and Building Act (1999).

The uncertainties possible in this kind of interview-based study are linked to the reliability and validity of the results. The method of snowball sampling can result in too positive a representation of the planning practices, because the key informants might recommend only the best-known, most qualified experts. In addition, those interviewed are not necessarily objective and unbiased when assessing the success of SEA and planning processes. However, the method was considered to reveal more information on common practices, especially the roles and practices of actors, than do selections of assessment reports and/or individual case studies from the more than 100 master plans prepared each year in Finland. Detailed description of the materials and methods, including the interview questions used, is given in *Article IV*.

5.5 Development of biodiversity impact assessment methodology

The fifth part of this work focuses on the development of conceptual criteria and a practical tool in response to the need for more holistic approaches to assessing impacts on biodiversity as expressed by the actors in the previous study (see *Article IV*). A need to increase communication and include actors' different views and values pertaining to planning and assessment in all phases of the assessment process was evident. In addition, the scant use of GIS-based

methods (see *articles I, II, and III*) coupled with the importance of GIS maps (see *Article IV*) highlighted the need for such a tool. Therefore, some kind of GIS-based tool giving room for creation of common knowledge/understanding and shaping choices in all phases of planning and ecological impact assessment was considered as one way to overcome several shortcomings in ecological impact assessment.

The conceptual sustainability choice space model developed by Haines-Young (2000) and Potschin and Haines-Young (2006, 2008) was taken as a starting point for development of ecological sustainability criteria, where the criteria were closely linked to economic and social criteria. Then, the ecological sustainability concept was broken down into two levels of ecosystem services criteria, and simple indicators concretising the criteria were developed. After this, indicator values were calculated on the basis of spatial information available throughout Finland in public databases and from the monitoring system of spatial structure (MSSS) (SYKE 2011). The criteria were designed especially for those middle-sized urban regions with 80,000–200,000 inhabitants facing the greatest land changes, threats to biodiversity, and needs for management and collaboration to maintain ecosystem services. The criteria and indicators developed were based on the Finnish national strategy on sustainable development and literature on ecosystem services, sustainability research, and best-practice principles of strategic environmental assessment (Bolund and Hunhammar 1999; de Groot et al. 2002; IAIA 2002; Pope et al. 2004; Kohti kestäviä valintoja 2006; Fischer 2007). The criteria and indicators were developed and initially tested in cooperation with actors in municipal local master planning in the cities of Lahti and Oulu. *Article V* details the methods applied in the development of ecosystem services criteria, including the list of criteria and indicators.

6 Results

The results of the five studies covering both EIA and SEA practices on a larger temporal scale, from the mid-1990s to late 2000s, as well

as methodological development motivated by them, are presented in this chapter. The results reveal the current knowledge basis as it is presented in EIAs and SEAs from the stage of the characteristics of the plan or project until that of the information provided on the planned monitoring. The results reveal the current structuring of impact assessment in terms of scoping, treatment of alternatives, and the influence of the assessment on the design of a plan or a project. In addition, the results show the roles of the actors in the assessment. Finally, after examining these elements, the chapter presents the ecosystem services criteria and their testing.

6.1 Knowledge basis in ecological impact assessment

6.1.1 The plan or project and its characteristics

The knowledge basis on which the decisions are made in environmental assessment appears to be weak. The weakness starts with lack of initial understanding of the characteristics of the project or plan and the environmental stress it could cause. In EIA, fewer than half of the ARs described the total area of development, meaning both the length and width of a linear development or the total area of a non-linear development. Most linear projects' ARs stated only the length of the construction (see *Article I*), which hampers assessment of the total area lost and the biophysical changes the project may cause. The environmental stress caused by the development was usually described qualitatively, and only rarely was more than one type of environmental stress affecting biodiversity quantified. In the Natura 2000 assessments, the level of information given on the plan or project was on a similar level (*Article III*; Söderman 2001, 2007). Parallel results have been reported from the other EU member states. A review examining 38 environmental impact statement (EIS) reports for road and railway projects, covering 1999–2003, from the UK, Sweden, France, and Ireland, found that the total length of the road or railway, though an elementary characteristic of the project, was not always specified in the EIS

(Gontier et al. 2006). The same deficiency in describing the length or total area was noted in a study of 40 EISs for road development projects from the UK between 1993 and 1997 (Byron et al. 2000) and earlier studies assessing a sample of 37 road EISs from the UK, covering 1990–1991 (Treweek et al. 1993), and a sample of 179 EISs, representing all development types, from the UK from 1988–1993 (Thompson et al. 1997). Furthermore, in reviewing 15 Finnish waste incineration ARs, covering 2001–2003, Jalava et al. (2010) found deficiencies in precise description of the project.

Information on the proposed development or plan is rarely presented in EIAs in such a way that decision-makers can identify the activities that might have impacts on biodiversity. Treweek (1996) expressed doubt as to whether the ecologists involved in ecological impact assessment actually are in possession of this information. It indeed appears that the ecologists have problems gaining relevant information on the planning. This was confirmed by interviews concerning local master planning, wherein ecologists mentioned difficulties in obtaining requisite information for the study (e.g., maps and aerial photographs of the planning area) (see *Article IV*).

6.1.2 The affected environment

The knowledge basis used as a baseline for ecological impact assessment does not cover biodiversity in terms of composition, structure, and key processes and their levels from genes to, for example, regional landscape level. The treatment of the baseline information does not cover different spatial or temporal scales and very rarely includes in-depth attempts to rank biodiversity elements by importance for further planning and assessment. Ecosystem services are not explored. The information provided in both ARs and local master planning studies concentrates on the species level and mainly on protected species and sites, with some general information added on vegetation and birds (see *articles I, II, and IV*). Although the flying squirrel is rather common in Finland, at the same time it is legally protected by the Habitats

Directive (CEC 1992, Annex IV(a) and Nature Conservation Act 1996) and has a special status. It is often the main reason for initiating ecological studies and for their level of detail (see *articles II and IV*), even a cause for other biodiversity elements being given short shrift. This emphasis on the species breakdown – as a significance and as a scalar dimension and a scalar-level choice – characterises ecological impact assessment. Connections between biodiversity levels are not considered. Information on land use outside protected areas is not gathered. In the UK, the national guidance requires describing the existing types of land use in the affected environment; therefore, the general land-use types in the affected area, including use for protected areas, are addressed, more or less, in the EISs (Treweek et al. 1993; Thompson et al. 1997; Byron et al. 2000). In contrast, Finnish EIAs do not include this, although the EIA Decree (1994, 1999, 2006) requires describing the affected environment.

For under half of the ARs it was clear that the ecological impact assessment had involved a new ecological survey. The surveys were mostly plant surveys but also included bird and fish surveys (see *Article I*). Many ARs did not provide any detailed information on the field surveys. Those that mentioned the duration of field surveys reported a period of one to three days, with only two exceptions, in excess of five days. In the Loviisa–Hikiä power-line EIA, the number of field days was 28, but this was not mentioned in the AR (see *Article II*). While this could suggest that there is more field work executed than reported, this is unlikely, because the interview of the developer revealed that studies for power-line EIAs' baseline are usually based only on map analysis, existing data from various sources, and some field verifications. The results of Jalava et al. (2010) also confirm that new studies or surveys are rarely conducted in Finnish EIAs and that most information presented is originally from earlier studies or reports probably meant for other purposes. In the appropriate assessments, field work was scarce, regardless of the fact that the entire assessment deals exclusively with species and habitat types. Only 12% of AA reports mentioned field studies

that lasted more than five days, and only a third mentioned short field visits, lasting one day or less (see *Article III*). In contrast, it appears that in land-use planning, the field work forms a much larger part of the work. The interviewees considered the field work to take one third to two thirds of the time spent on the whole study, including, in addition, a survey of written material, maps, aerial photographs, existing studies, and databases (see *Article IV*).

Usually the field surveys were carried out at the right time of the year. Ecological data were not localised in sufficient detail, either in ARs or in AA reports. In addition, the data were presented mostly on presence/absence level, without any quantitative account of the habitat area or species abundance information (see *articles I and III*) (Söderman 2001, 2007). The simultaneous scarcity of up-to-date field surveys, map presentations, and land-use information suggests, when taken in conjunction with the concentration on protected species and sites, that the knowledge on which the impact prediction should be based is not sufficient for any sound analysis in the impact prediction. Similarities and differences in relation to other EU member states exist. In the UK, most surveys between 1993 and 1997 concentrated on broad categories of habitat and vegetation in their mapping (Byron 2000), and the second most frequent survey type involved a particular species group (the badger). The number of field surveys in 1993–1997 (Byron et al. 2000) was nearly two times larger in the UK than in Finland, but the Finnish quantity was around the same as earlier in the UK, in 1990–1991 (Treweek et al. 1993) and 1998–1993 (Thompson et al. 1997). However, Byron et al. (2000) are sceptical about the benefits of a larger number of surveys, because the quality of impact prediction was not improved simultaneously and so the studies were not of an appropriate type to capture relevant ecological information. Sparseness of new surveys and attention mainly to single species on a presence/absence level that emphasises protected species and lacks an ecosystem or even broader biodiversity focus in EIAs have been documented in Sweden, France and Ireland (Gontier et al. 2006), the

USA (Southerland 1995; Atkinson et al. 2001), Australia (Warnken and Buckley 1998), and Israel (Mandelik et al. 2005a) as well as in less developed countries such as Sri Lanka (Samarakoon and Rowan 2008) and India (Khera and Kumar 2010). An interesting justification for species-level focus was reported in Atkinson et al.'s 2001 study of 35 EISs covering 1993–1998. They noticed that many EISs stated that biodiversity in a broader sense is not an issue on the project level but should be analysed at a strategic level. This resembles the views of the planners in local master planning that broader biodiversity issues, such as ecological networks, belong to regional planning (see *Article IV*). However, the emphasis on species level in local municipal planning is in contrast with the results of a review of ecological impact assessment for five spatial plans from the 2000s in the Netherlands, of which two were local plans. In all of these plans, the assessment looked predominantly at changes on the ecosystem level by means of high-quality GIS maps. In only in a few cases were protected species, flora, and fauna considered in SEA (Kolhoff and Slootweg 2005).

There were severe shortcomings in description of methods, data, and uncertainties attached to them. Very few ARs and AA reports describing uncertainties evaluated these uncertainties' effects with respect to use in impact prediction (see *articles I and III*) (Söderman 2001, 2007). Parallel results have been found from the UK, Sweden, France, and Ireland (Gontier et al. 2006), where it was impossible to judge what methods had been utilised in the baseline studies for half of the EISs.

6.1.3 Effects on biodiversity

With an inadequate or distorted knowledge basis about biodiversity, meaningful impact prediction is more or less destined to fail. The results of all four studies indicate that this is usually the case (*articles I, II, III, and IV*). In the impact predictions, both in ARs and in AA reports, mainly qualitative direct impacts on species level were dealt with, but even as such they were vague, without much quantification.

The situation for prediction of impacts on biodiversity is about the same all over Europe (Byron et al. 2000; Gontier et al. 2006) and the world (e.g., Warnken and Buckley 1998; Atkinson et al. 2000; Samarakoon and Rowan 2007; Khera and Kumar 2008). This applies to not only ecological impact assessment but the quality of all impact assessment in Finnish ARs. Jalava et al. (2010) reported also that the predictions in their sample ARs were rather evasive. Rare quantification of impacts mentioned loss of habitats of single species in proportional terms, which cannot be considered good practice, because the impacts can be diluted by scaling (João 2002); also absolute values and location of impacts should be given. Proportional values are meaningless in the absence of knowledge of where the impacts are taking place in relation to the most representative habitat types on the whole Natura 2000 site. Often, indirect impacts were mentioned in ARs only as 'indirect impacts', without specification of their nature. The AA reports listed many indirect impacts. The time scale or duration of the impacts was rarely mentioned in ARs and AA reports; for the most part, these mentioned whether the impact was temporary or permanent. In the Loviisa–Hikiä EIA case, the impact prediction concentrated on the flying squirrel, with respect to which the impact assessment was detailed and sufficient. However, the rest of the impact assessment remained vague. In connection with local master planning, a surprising result was that planners did not consider biodiversity an important issue in the impact prediction phase as long as data and land-use recommendations from the baseline studies would be taken into account in the planning (see *Article IV*). However, the minor role of ecological impact prediction was criticised by the environmental authorities.

Assessment methods were rarely mentioned and included expert interviews, photography techniques for examining visual impacts, comparison matrices, and ecological environment information classification. No more than half of the EISs and AA reports there used maps to present the baseline results, and only one third presented some ecological impacts via maps (see *articles I and III*) (Söderman 2001, 2007).

Except for presenting the results, GIS methods were not used – even the simplest overlaying techniques (see *articles I, II, and III*) (Söderman 2001, 2007). This was also reported by Gontier et al. (2006) for the UK, Ireland, Sweden, and France, where GIS techniques were used only to display and mapping functions in EISs. The sketchy descriptions of both baseline study and impact prediction methods and their uncertainties hampers the interpretation of the results in decision-making (see *articles I, II, III, and IV*). This is consistent with the findings of Therivel and Ross (2007) on consultants’ unwillingness to present anything that will put their reputation at risk. However, leaving the methods out of the documentation makes impact prediction appear to be a ‘black-box’ exercise or, even worse, just an exercise in guesswork.

6.1.4 Cumulative impacts

Only a small minority of ARs addressed cumulative impacts. The receptors affected – the valued ecosystem components – were not identified. Some mentions existed of possibly cumulative barrier effects caused by, for example, a railroad and road together but without attempts to assess them in more detail. Cumulative impacts were not addressed at all in the Loviisa–Hikiä power-line EIA, because the developer and the EIA authority deemed this to be too difficult a task and beyond the concern of a single project impact assessment (see *Article II*). Parallel results have been reported by Wärnbäck and Hilding-Rydevik (2009), who found in an interview study of 10 Swedish EIA and SEA actors that these were unaware that the EIA and SEA directives require cumulative impact assessment. It was the perception of the Swedish actors that the national legislation did not require addressing cumulative impacts. This might be the case also with respect to the Finnish EIA legislation, where the requirement of cumulative effects’ assessment is hidden in the definition of environmental impacts in Section 2e of the EIA Act (1999), referring to ‘interaction between the factors’. Although the whole Natura 2000 assessment exercise is based on either individual impacts or impacts in com-

ination with other projects and plans, only 15–29% of AA reports mentioned cumulative impacts and cumulative impact assessment was not addressed at all on the species or habitat type level (see *Article III*). Poorly performed assessments of cumulative impacts have been reported in both EIA and SEA in the UK (Cooper and Sheate 2002; Therivel and Ross 2007), the USA (Burriss and Canter 1997), and Canada (Bonnell and Storey 2000). In the UK, a study of 50 EISs from 1989 to 2000 showed that under half of the EISs addressed cumulative impacts and only eight EISs provided analysis of cumulative impacts (Cooper and Sheate 2002). Three of these EISs were associated with appropriate assessments of impacts on Natura 2000 sites, and the rest with other nationally or internationally protected sites. This parallels Finnish practice, in which cumulative impacts are more often assessed in AA reports than in ARs (see *articles I, II, and III*).

6.1.5 Significance

The significance of the impacts was evaluated in fewer than half of the ARs (see *Article I*). Only a small proportion of the ARs that addressed significance presented how it had been determined. However, doing so was of little additional value, because impacts were presented as significant, moderate, minor, or negligible without further definition of the criteria used. In ARs, only the magnitude of an impact and the protection status of the receiving environment were mentioned as significance criteria (see *Article I*). These results correlate with those from the UK in 1988–1997, except that significance criteria were given more often in the UK than in Finland (Trewick et al. 1993; Thompson et al. 1997; Byron et al. 2000). In a study of 30 EISs from the UK looking at more recent years, 2000–2006, that dealt with the treatment of significance of visual and noise impacts, Wood (2008) found many approaches to defining significance classifications. Still, a third of the EISs did not attempt to communicate the approach employed to evaluate visual impacts. The criteria used most often in examination of visual impacts were landscape sensitivity and

magnitude of change, while for noise impacts the main criterion was relative or absolute change in decibel values. Significance of cumulative impacts was not addressed in any ARs in the sample (see *Article I*). In the Loviisa–Hikiä power-line EIA, the assessment of impact significance was based on expert opinion stating a threshold value for flying squirrels’ distance from the power line before their habitats would be disturbed (see *Article II*). In the Natura 2000 assessment procedure, absence of significant adverse impacts on Natura 2000 sites is a prerequisite for proceeding with the project or plan in permission and planning processes. Accordingly, the significance of impacts was addressed in nearly all AA reports (see *Article III*). Usually qualitative reasons for significant deterioration of certain species statuses or habitat types were given. Attempts were made to classify the significance, from severe negative to highly positive effects, but without explanation of the criteria for these classes. Only two out of 73 reports explained the criteria (Söderman 2001, 2007). The issue surrounding transparency in evaluation of impact significance appears to be a perennial one: a large proportion of even relatively recently prepared EISs still fail to explain the approach used to evaluate and communicate on judgements regarding impact significance (Wood 2008).

6.1.6 Mitigation and monitoring

Need for mitigation is widely recognised in Finnish ecological impact assessment practice (see *articles I, II, and III*). It was mentioned in the majority of ARs and AA reports, and details were often provided. In contrast to results obtained from the UK and some other countries (Thompson et al. 1997; Byron et al. 2000; Mandelik et al. 2005a), no cosmetic measures such as tree-planting or landscaping have been proposed as major mitigation measures. Usually they were not even mentioned as ecological mitigation measures. Especially in AA reports, some mitigation measures were addressed in great detail – with, for example, proposals as to how far certain activities should be moved from habitats of certain species. Environmental

authorities paid special attention to mitigation measures in their statements on significance of effects. The implementation of mitigation measures was a precondition for over one third of the projects’ plans for not causing significant adverse effects (see *Article III*). However, there were also inadequacies. All ARs and many AA reports did not specify which impacts certain proposed measures were designed to mitigate and evaluation of measures’ effectiveness was not even attempted (see *Article I*) (Söderman 2001, 2007). Considerable effort was also employed on mitigating effects for the flying squirrel with respect to the Loviisa–Hikiä power line with route bends (see *Article II*).

Monitoring was a poorly addressed issue in all types of ecological impact assessment (see *articles I, II, III, and IV*). In the case of the Loviisa–Hikiä EIA, the requirement for monitoring was passed over with a note that the proponent participates in research projects studying impacts of power lines at a general level. However, this does not correspond to the requirement for a specific monitoring scheme as defined by the EIA Decree (1994, 1999). The monitoring scheme requirement was followed to a certain extent in ARs that made proposals concerning species-level monitoring, but it was without specification of the performer or the period. In comparison to EISs from the UK (Thompson et al. 1997; Byron et al. 2000), monitoring was addressed more often and in greater detail in Finland. The more frequent addressing of monitoring may be caused by the explicit requirement in the EIA Decree (1999). Nevertheless, according to the results of Jalava et al. (2010), the present requirement regarding presentation of a monitoring scheme without actual obligation or commitment to monitoring was seen by EIA authorities and consultants as meaningless and unnecessary in practice. For land-use planning, the Land Use and Building Act (1999) does not require monitoring of individual plans, and, hence, no monitoring activities are planned for impacts of local master plans. Monitoring is seen as covered by general monitoring of the environment. The results concerning mitigation and monitoring, especially in EIA and Natura 2000 assessment, along with

a proactive stand on impact prediction in local master planning (see *articles I, III, and IV*), indicate a strong interest in application of the precautionary principle, even without proper information on biodiversity elements that might be affected and impacts on these. This raises the question of whether the monitoring and mitigation activities address the most significant impacts or merely the most obvious ones.

6.1.7 Changes over time

The study of ARs explored whether the Nature Conservation Act (1996) had changed the quality of ARs measured by EBIs. It had not. The average index was the same for all three ranges of years: 1995–1996, 1997–1998, and 1999–2001 (see *Article I*). In Natura 2000 assessment, there was an improvement in the quality of AR reports from the 1997–mid-2001 period to the period mid-2001 to 2005 for those reports measured by both NI45 and NI21 indices. The improvement was greater when measured by the index covering the most important questions, suggesting that the focus became clearer. In addition, the percentage of deficient reports decreased and of the highest quality increased. As to the individual parts of the AA reports, there were improvements in the description of the plan or project and its environmental stress, individual species and habitat types, and the details of mitigation measures. The localisation of information on maps deteriorated, which may indicate deterioration of the information basis as well. Without clearly defined biodiversity elements to study, ecological impact assessment practices appear in EIA and SEA, by contrast to Natura 2000 assessments with habitat types and overall integrity considered also, to suffer from sluggishness in adopting a broader understanding and treatment of biodiversity (see *articles I, II, and IV*). While it should be noted that my EIA data extend to the year 2001 while Natura 2000 assessment data extend to 2005, the actor views from 2007 represent a relatively recent situation.

6.1.8 Reporting

Impact assessment documents appear to have a dual role in knowledge-sharing. Firstly, the documents are produced as a quality check for the authorities. In this respect, there appear to be deficiencies. Some aspects of important information are omitted from both the AP and the AR (see *Article II*). Also omitted are many features that are important for enabling assessment of impacts' significance, such as methods (see *articles I, II, III, and IV*). Surprisingly, regardless of the obvious omissions, the reports appear to meet the needs of EIA authorities (see *Article II*). This might be due to informal negotiations between EIA authorities and developers (see *Article II*) (Pölonen et al. 2011) that are based on knowledge not actually reported in the documents. By contrast, when considering the significance of impacts on Natura 2000 sites, the authorities regarded 22% of the AA reports as inadequate. When the authorities added their own information, in some cases it was possible to evaluate the significance of impacts, but still in the majority of these cases, where inadequate information was provided by the AA report, the authorities considered it impossible to evaluate the significance of the impacts. Secondly, the reports are produced for communication and public participation. In the Loviisa–Hikiä EIA case, the proponent and the main consultant saw the main function of the reports as being to provide information to the public; therefore, there was a focus on keeping the reports short and concise. Brevity of reports is not in itself a problem, but a transparency issue arises when reports omit central information on fundamental choices made in the planning and decision-making, thus preventing the public and stakeholders from engaging in the assessment process and contesting the knowledge about biodiversity and ecosystem services.

6.2 Structuring of the ecological impact assessment process in EIA, Natura 2000 assessment, and local master planning

6.2.1 Scoping of the ecological impact assessment

Inadequate scoping is one of the main reasons behind the deficiencies of knowledge production in ecological impact assessment. Already since the late 1980s, it has been argued that many of the problems with unsatisfactory EIA studies are associated with a lack of sound scoping (Morgan 1988; Kennedy and Ross 1992). Without specifying the area affected, one cannot proceed to meaningful baseline studies and impact prediction. Under half of the ARs or AA reports indicated attempts to define spatial boundaries in the form of an affected area (see *Article I*) (Söderman 2001, 2007). The most usual delineation of the affected area was a 100-metre-to-two-kilometre-wide corridor for linear developments or other distance-based lineation for non-linear projects. In the Loviisa–Hikiä power-line EIA, the area studied was a corridor 100–200 metres wide (see *Article II*). This suggests that scale issues in terms of the areas influenced are not manipulated to ‘dilute’ the impacts. On the contrary, the area influenced might be demarcated as too limited to capture all impacts of the project. A clear deficit is that the definition of the study area is determined mainly on a non-ecological basis. Spatial or temporal (or any other) boundaries allowing a ‘bigger picture’ for cumulative impact assessment were not defined in the majority of EIAs and Natura 2000 assessments (see *articles I, II, and III*) (Söderman 2001, 2007). This is an important finding, especially with respect to Natura 2000 assessment, in which even the screening phase should cover identification of cumulative effects from other projects and plans. Without any consideration of the area from which the effects might come and where other projects and plans and other activities and their impacts should be looked for, it is difficult to capture any cumulative effects. Except in the odd AA report, boundaries having to do with

time were not dealt with; this includes specification of the time range for which the impact years are going to be predicted (see *Article III*). Also not addressed were the limitations and opportunities created by the amount of time reserved in EIA for the baseline studies and impact assessment, with account taken of the fact that a full-blown EIA takes 14 months on average (Pölonen et al. 2011).

Selection of the biodiversity elements, VECs, to study in baseline surveys and consequently in impact prediction was done superficially or not at all. It was impossible to conclude from any of the ARs studied whether the biodiversity elements for detailed analysis had been intentionally selected. Though the indirect information given in the reports suggested that the biodiversity elements, usually only species, were chosen on some basis, the reasoning applied in this was not reported transparently. In the Loviisa–Hikiä power-line EIA case, the developer gave the main consultant and sub-consultant a list of biodiversity issues for study that covered the protected species, protected areas, and areas identified in national inventories and protection programmes as worthy of protection (see *Article II*). The list was used only to some extent in practice. Only the flying squirrel was mentioned in the AP, and why this particular species was chosen for detailed studies was not mentioned. It emerged in the interview of the developer that the legal protection of this species was the main reason. In general, both ecological scoping of EIA and the work in local master planning are strongly driven by motivation to avoid legal problems in decision-making in permit and approval procedures instead of by a desire for broader maintenance of biodiversity and ecosystem services (see *articles II and IV*).

Since the methods for baseline studies are not addressed sufficiently and hardly any methods for impact prediction are addressed at all in ARs and AA reports, this suggests that they are not properly planned and reported in the scoping phase either (see *articles I, II, and III*). The AP for the Loviisa–Hikiä power line addressed only the flying squirrel inventory methods, excluding impact prediction methods. Any other baseline methods mentioned were just field checks. This

was the situation in the beginning of the assessment as well. Thus reporting corresponded to the existing level of scoping. The interviews of ecological impact assessment actors confirmed the vagueness of scoping. The normal scoping procedure is to commission a biodiversity study with a very open and imprecise assignment, including a list of the most obvious legally protected biodiversity elements or, alternatively, a list of species so long that they cannot be studied within the limits of the time and money allocated for this (see *Article IV*).

6.2.2 Dealing with alternatives

Alternative options were usually dealt with in EIA practice. The zero alternative was not used as justification for a project (see *Article I*). The case study of Loviisa–Hikiä demonstrated that biodiversity considerations affect the selection of alternatives. Those alternatives having the most severe impacts, at least on legally protected Natura 2000 areas, were excluded from the beginning (see *Article II*). On the assumption that this would be the prevailing practice, the most harmful plans and project alternatives were screened out as infeasible and not passed on to appropriate assessment at all. However, this assumption does not hold. The study of AAs demonstrated that almost a fifth of projects and plans end up being declared infeasible because of significant adverse effects on Natura 2000 sites (see *Article III*). One assessment reached this conclusion after three rounds of assessment because there was only one predefined option. The Habitats Directive (CEC 1992) does not require treatment of alternatives in AA. However, it has been considered good practice in Finland to include alternatives in the procedure, to enable choice of the least harmful alternative as a mitigation measure (Söderman 2003). This practice appears to be working to some extent, as 70% of the projects and plans were considered to be feasible in combination with mitigation measures and these mitigation measures also included choice of the least harmful plan alternative either for a local master plan or for a detailed plan or an alternative route for linear projects (see *Article III*).

6.2.3 The potential influence of ecological impact assessment in planning

It is often concluded that EIAs' contribution to project design is rather modest (Wood 2003; Cashmore et al. 2004; Jay et al. 2007). However, their contribution depends on how permission processes are linked to EIAs. Pölonen et al. (2011) discussed the problems in linking EIA and decision-making and suggested that one of the main problems is that the permit authorities are not legally obliged to follow the recommendations, but the results of AR review and a case study (see *articles I and II*) indicate that the results and recommendations communicated in the main EIA documents did not enable comprehensive consideration of biodiversity even if they had to be heeded, because the knowledge basis they provided was so incomplete. If the documents do not say anything relevant about the biodiversity elements or impacts on them, they cannot contribute to the project design. Poor documentation can also be an indication that the whole EIA process has been undertaken as a separate, 'add on' exercise, an extra burden without there being real willingness to use EIA as a procedural tool to guide the project design. Natura 2000 assessment with its greater legal force did not appear to perform much better. However, its objective is not to advance consideration of impacts on biological diversity in general as EIA should (Act on Environmental Impact Assessment Procedure 2006, sections 1 and 2) but to prevent significant adverse effects on predefined conservation objectives. Therefore, the focus of the assessment can be narrower. However, in practice, this narrow focus has been interpreted somewhat too narrowly and thus as not extending to the whole Natura 2000 site and its integrity, with consideration of factors supporting the ecological character and preconditions for the existence of individual species and habitat types (see *Article III*). There are signs that Natura 2000 appropriate assessment procedure can affect project design. Accordingly, the obligation to take the results of the assessment into account appears to have an effect.

6.3 Actors and their roles in ecological impact assessment

The results of interviews of local municipal planning actors (in *Article IV*) demonstrate that actors have different perceptions of the purpose and content of ecological impact assessment and that their perceptions mirror their different concepts of biodiversity. Although they represent only the views of actors in local master planning, in a small country such as Finland, the authorities (previously regional environment centres and now the ELY Centres) and ecology consultants have been more or less the same in EIA and local master planning. Therefore, these actors can represent actor views also in EIA and Natura 2000 assessment. By contrast, planners represent actors different from project proponents. However, the environmental authorities, ecology consultants, and planners represent only a part of the field of actors or those involved in ecological impact assessment (see Table 8, on page 40) and the results must be interpreted in view of this.

In practice, among the most pressing problems expressed by the actors was the absence of joint effort to set spatial and substantive boundaries to the assessment exercise. Although the planning area is usually smaller than the affected area, impact areas tend to be delineated to be the same as the planning areas. In addition, some planners, along with the authorities, stressed the need to survey biodiversity elements in the areas where the land-use changes are the greatest, while other planners emphasised the need to survey biodiversity elements in areas that will be left outside the development. This indicates that some planners still take the traditional nature conservation view of just leaving some elements outside the development while others hold a more holistic view of biodiversity even in the scoping phase. The Finnish environmental authorities guiding the process had the most comprehensive view of biodiversity elements, and they criticised omission of ecological connectivity and larger biodiversity units, such as ecosystems and green infrastructure, in local-level land-use planning (see *Article IV*). Some planners still had a very narrow concept of bio-

diversity while others demanded consideration of larger units also and called for more holistic approaches. It appears that there is a positive tendency arising in land-use SEA – toward a holistic view. Nevertheless, the holistic view does not appear to be concretised in planning practices.

The absence of interaction between planners and consultants in scoping was perceived as a problem. Although formal and informal negotiations were held between planners and authorities in the scoping phase, the ecologists were not part of these. When they enter the process, the time and monetary resources have already been decided upon and thus have determined the content and methodological choices for the baseline studies to be carried out by the ecologists and for further impact prediction. The authorities have called for better use of the participation and assessment schemes for negotiations among all core actors and stakeholders. In the present practice, stakeholders do not have any role in the ecological impact assessment.

The situation for EIA appears to be, in at least in some cases, slightly different: the proponent and EIA authority too make scoping decisions (see *Article II*). However, biodiversity experts with the EIA authority are not always directly involved in the scoping phase, so scoping decisions may not be based on sufficient ecological expertise. Stakeholder involvement and participation in the Finnish EIA system enables interaction among proponents, authorities, and stakeholders (Pölonen et al. 2011). However, in a top-down participatory tool such as EIA, the developer is capable of influencing the arenas, value choices, timetables, and agenda of the stakeholder involvement (Morgan 1988). At the public hearings of the Loviisa–Hikiä EIA, held in both the scoping and the assessment report preparation phase, local inhabitants were not greatly concerned with biodiversity issues, and they argued that the flying squirrel issue was getting too much attention. However, in public hearings in the scoping phase, local stakeholders raised the need to survey locally important bird sites. These were not studied, because they were not considered ornithologically valuable, but some information on bird interactions with

power lines was provided (see *Article II*). This was a good illustration that stakeholders can have very different views of significance than the core actors in the ecological impact assessment do.

In baseline studies, both project proponents and land-use planners appear to rely heavily on the expertise of the ecology consultants, and ecologists are used widely in EIA and local land-use planning (see *articles I, II, and IV*). One third of the ARs and, surprisingly, also of AA reports did not address the use of a professional ecologist, and it remained unclear whether a qualified ecologist was involved. In a parallel study of EIA in Israel, a country that has a relatively small community of professional ecologists involved in ecological impact assessment, as does Finland, it was found that 60% of the EIAs involved an ecologist and this involvement was the second most influential factor in determining the quality of the EIS report, after quality of scoping (Mandelik et al. 2005a). From this and results from the interviews of local master planning actors (see *Article IV*), it can be concluded that important decisions are made in the scoping phase and that if they are incorrect, things can still be rectified later with the use of a qualified ecologist, within the limits of the allotted time and monetary resources. However, if the consultant selected has a narrow area of expertise and specific biodiversity elements have not been selected as VECs in the scoping stage, the possibilities for production of meaningful information are limited. Unquestionably, the best phase for collaboration of all actors and involvement of ecology consultants, especially if there are not biodiversity experts available or involved within the guiding authority, is the scoping phase.

In local municipal planning, the impact prediction and the subsequent phases are usually carried out by the planner or by a consultant different from the one involved in the baseline study phase (see *Article IV*). The important finding was that the majority of authorities and consultants called for more collaboration while planners were satisfied with the present practices and had a more positive impression of the use

of the baseline studies in the planning process when compared to authorities and consultants.

At present, the role of EIA authorities or environmental authorities in Natura 2000 assessment and local land-use planning is directed toward substantive and procedural quality assurance for the ecological assessment process (see *articles II, III, and IV*). My results related to the ecological impact assessment process and results pertaining to the EIA process as a whole (Pölonen et al. 2011; Jantunen and Hokkanen 2010, 2011) point to authorities' formal statements as having a decisive role in the process. This is emphasised in the Natura 2000 assessment process in consequence of its binding nature for the final formal permit decision. In comparison of the views of significant effects, the AA reports prepared by developers, planners, or their consultants tended to regard the adverse effects as insignificant more often than did the statements given on them by the authorities (see *Article III*). In fact, the AA report and the official statement by the authority were originally in accordance in under a tenth of the cases as to impact significance. The final decisions of the competent authorities or actual realisation of mitigation measures has not been studied, but it may be indirectly concluded from the quantity and content of second-round or even third-round AAs and statements given on them that the competent authorities return at least some of the projects and plans that have inadequate AA reports and that demonstrate significant adverse effects, sending them back to the assessment process. The number of returned projects and plans is half that of the projects and plans regarded as infeasible or inadequate by the authorities (see *Article II*) (Söderman 2007).

The group of ecology consultants carrying out ecological impact assessment is diverse, including ecologists, architects, and also experts without ecological expertise (see *Article IV*). All actor groups in local master planning who were interviewed regarded a tendering procedure that emphasises costs instead of content or quality as lowering the quality of assessment. Problems with quality were seen as related not to the professional skills of individual ecologists but to the structuring of the whole planning

process. For example, it is not beneficial for the integration of planning and the ecological impact assessment process that baseline studies and impact predictions are carried out by totally different experts. In this practice, an integrated or decision-making and planning-centred environmental assessment process is not feasible (Slootweg et al. 2006; Partidário 2007). The results of the interviews confirm earlier results of a study exploring possibilities for certification of ecologists in the field of ecological impact assessment, which concluded that problems in the quality of ecological impact assessment cannot be resolved by intervening only in terms of professional standards, and that the problems are much more complex (Söderman 2004). In addition, professional standards are not well attuned to the real world of contested values and rationales, wherein decisions made by the actors on the form and content of impact assessment are inevitably value-based and need to be negotiated in each specific planning and decision-making situation (Richardson 2005).

Shortcomings in collaboration between actors and stakeholders in all types of ecological impact assessment are evident in both EIA and local master planning SEA. Today's practices in ecological impact assessment in Finland appear to follow the simplest level of knowledge-sharing, in the form of a very expert-driven, one-way information provision approach, not reaching the full potential of engagement, not to mention building the capacity mentioned by Sheate and Partidário (2010) as inherent in knowledge brokerage approaches (see *articles II and IV*).

6.4 Promotion of ecological sustainability through ecosystem services criteria

6.4.1 Ecosystem services criteria and indicators

Ecosystem services were operationalised for practical planning and impact assessment purposes through the development of ecosystem services criteria and indicators. Further detail concerning the criterion and indicator develop-

ment presented below can be found in *Article V*. Ecological sustainability was interpreted as long-term functionality of ecosystem services. The criteria and indicators were directed to land-use planners and ecological impact assessment practitioners in middle-sized urban regions of 80,000 to 200,000 inhabitants but are usable throughout the country, from municipal planning (performed by municipalities) to provincial planning (performed by the regional councils).

The criteria and indicators' development was based on the conceptual model of sustainability choice space, according to which stakeholders, in accordance with their values, determine a feasible ambition level for biodiversity goals (Opdam et al. 2006; Potschin and Haines-Young 2006, 2008). Then planning and environmental impact assessment strive to find a landscape design appropriate for reaching the goals. The sustainability choice space consists of land-use choices that are considered sustainable. Within the limits of the space, the planning choices sustain the desired ecosystem services. Outside these limits, the capacity to supply services is lost (see Figure 6). Ecosystem service indicators are used to set the limits for these planning choices, with the limits determined both by biophysical features (biodiversity or generating units of ecosystem services, including soil, water, flora, and fauna) and by stakeholder values. These limits may vary in time according to changes in scientific information, technology, stakeholder values, risks and uncertainties, and benefits.

In the development of criteria and indicators, ecological sustainability was deconstructed into comprehensible pieces by means of two-level criteria. The main criteria deal with the state and use of ecosystem services and with threats to ecosystem services. The five main groups of criteria address land use, green infrastructure, recreation, the water cycle, and the transport system. The 17 second-order criteria concretise ecological sustainability objectives in more detail and present the targets against which one evaluates whether the goals of the main criteria are met. The indicator values specify what is considered by stakeholders (supported by information on biophysical features of the environ-

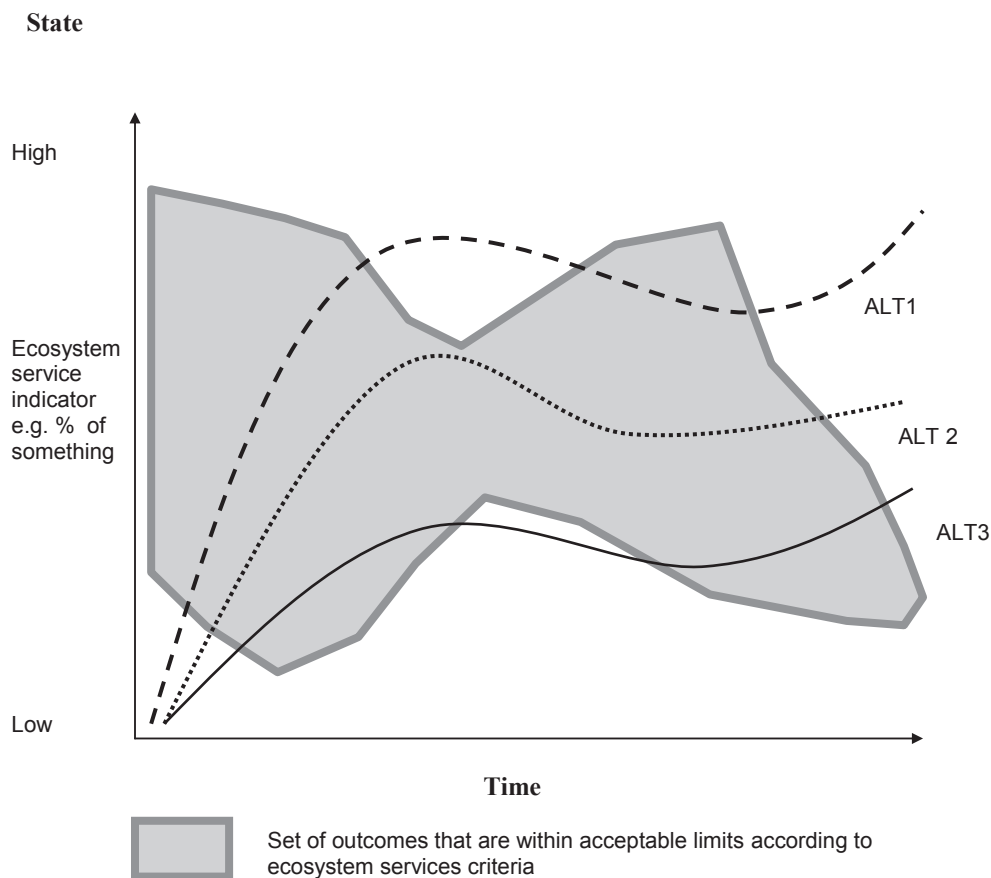


Figure 6. Conceptual model of sustainability choice space. Modified from Potschin and Haines-Young (2006).

ment) to be a desirable level of a certain indicator for provision of the desired ecosystem services. Directly or indirectly, the indicators represent all ecosystem services identified in urban regions in the international ecosystem services literature reviewed by Niemelä et al. (2010).

In addition to their use in target-setting in the early stages of planning, the indicators can be used for describing the baseline situation; assessing and comparing the impacts of a certain project, plan, or programme; and monitoring impacts in view of the targets set, the baseline, or predicted impacts. Although the criteria and indicators were developed for land-use planning in urban regions (SEA), they can be used in a number of EIAs as well. Then the study area will affect the choice and use of criteria and indicators. Some indicators work only in

an urban region or larger spatial units (e.g., the indicator addressing carbon sinks), while some are also applicable at a very detailed level of planning (e.g., the indicator addressing accessibility of nearby recreation areas).

The indicators are expressed as 1) simple quantitative ratios and proportional values (e.g., the free shoreline in proportion to the total amount of shoreline), 2) tables describing the total amount of certain features (e.g., kilometres of free and built-up shoreline), and 3) maps describing the spatial distribution of these features (e.g., where free and built-up shore areas are situated). The quantitative ratios enable comparison between areas and allow broad target-setting. To rule out dilution by scaling (João 2002), indicators are expressed also as absolute values. The simplicity of the indicators makes them easy to use in participatory and col-

laborative planning situations involving stakeholders. All indices with weighting of variables were intentionally avoided as ‘black boxes’ in planning situations. The indicators were kept as simple as possible but numerous, providing the possibility of picking the most useful ones for the planning problem at hand and to suit the availability of data. Several indicators can be analysed and displayed simultaneously as overlaid GIS analyses and presentations. Indicators are presented as quantitative ratios that vary with the area being studied: a municipality, an urban region, the area of a regional council (province), or the affected area in the case of EIA. In pilot studies, extension of the functional urban areas by concentric zones of 10 and a further 15 kilometres’ width was used.

6.4.2 Development and testing of the criteria and indicators

The development work was done in 2008–2011 by an ecosystem services research team consisting of geographers, ecologists, and GIS experts at the Finnish Environment Institute, the Department of Environmental Sciences of the University of Helsinki, and the consultancy SITO, and it was designed and led by the author of this thesis. The team developed criteria based on the international and national literature on ecosystem services and discussions in internal workshops. Since the aim was to demonstrate ecosystem services spatially, one of the main starting points was availability of spatial data. Two sustainability development workshops were held, where users – potential and some already with experience – discussed the use of criteria and indicators. The criteria and indicators were tested in three pilot planning situations: an ongoing local master planning process for Lahti, possible renewal of the Oulu region’s joint local master plan, and monitoring of a regional development programme by the Päijät-Häme regional council. The original plan was to test the criteria in real ongoing region-level strategic planning processes such as joint regional land-use or transport plans, but, of the five urban regions contacted, only Lahti and Oulu expressed interest in involvement in the

development and testing of the criteria. Furthermore, it emerged as the development project progressed that the interest in the Lahti region concerned mainly land-use planning of a single municipality and political changes linked to the administrative merging of municipalities caused the original boundaries for regional co-operation in the Oulu region and thus the delineation of the study area to change. However, from the testing perspective, this was not necessarily a disadvantage, because it showed the administrative reality in which criteria are used in real life and experiences of different planning situations were obtained. The Päijät-Häme regional council offered a third (rather small) pilot case in the form of the need for indicators for monitoring the actualisation of goal-oriented trends defined by its regional development programme. Their goal-oriented trends resembled the second-order criteria. For the Lahti and Oulu urban regions, the ecosystem services research team calculated the indicator values and produced the maps covering the study area. In Oulu, an additional set of indicator results was produced from the administrative area exactly following the borders of the 10 municipalities, by the request of the Oulu regional project group, who felt that the function boundaries were not appropriate for the joint master planning. In Lahti, an additional set of results, related to the indicators chosen by the Lahti project group, was produced in impact assessment for three alternatives in the local master plan for Lahti. In this, the study area was limited to within the borders of the city of Lahti. In Päijät-Häme, the testing considered only selection of the most suitable indicators. The council plans to produce the indicator analyses itself in 2012.

In view of the testing, the numbers of criteria and indicators were changed between the first sustainable development seminar, held in June 2009, and the final one, in February 2011, as follows:

Year	ES criteria	Second-order ES criteria	ES indicators
2009	12	47	110
2011	5	17	28

Firstly, overlapping and non-comprehensible indicators were removed. After this, data availability was the main factor in inclusion. Although the indicators had been designed from the data availability perspective, it was surprising how often data problems were encountered: national databases were inconsistent, they did not cover areas consistently throughout the country, and time series were not exhaustive. For ecosystem-services-linked spatial data, the most useful proved to be the CORINE Land Cover data (EEA 2009), available in a detailed Finnish version only for years since 2000, and still data conversions and new analyses were needed for calculation of the relatively simple indicators. The data challenges were even greater with municipal data. When there was a request for data to utilise for those indicators that could not be calculated from the national data, both the researchers and the members of the regional project groups anticipated that the municipal data would be rather easily deliverable and the researchers would be able to carry out analyses. However, it became apparent that municipal spatial data seldom existed. When such data did exist, the material had been collected in accordance with a variety of concepts and methods in the municipalities of the regions and the coordinate systems and GIS formats used also varied. As a consequence, some second-order criteria and indicators related to issues such as outdoor recreation areas, silent areas, land-extraction sites, and water quality had to be abandoned.

Several lessons were learnt through development and testing of the ecosystem services criteria, which are expressed here as opportunities and problems, along with ways to overcome the latter.

Opportunities

The development work was useful for integration with the real ongoing planning situations or situations resembling them. The connection to practical planning and close collaboration throughout the development work with regional project groups in the form of commenting upon the evolving versions of the criteria and indicators eliminated the most theory-bound and im-

practical ideas. However, these ideas nonetheless were presented, discussed, and tried with existing data before elimination. Therefore, the full potential provided by ecosystem services research was actually tested. Without the constant communication between the land-use practitioners and the researchers, the criteria would have remained much less user-friendly.

Availability of spatial data worked well as one starting point for the development work. It guaranteed that a spatial approach, necessary for addressing spatially bound ecosystem services and biodiversity, was prioritised. It also provided opportunities to find use for the existing spatial data from the ecosystem services and biodiversity perspective. It broadened the use of the existing data instead of finding new needs to develop resource-intensive data collection systems. For example, most land-use indicators for the second-order criterion 'community structure is consolidated' describe the desired levels of threat to ecosystem services. Therefore, use of the Finnish monitoring system of spatial structure, MSSS (SYKE 2011), data to address the ecosystem services criteria can actually reveal something important about the land-use choices that could maintain biodiversity and ecosystem services.

Taking a very broad ecosystem services approach that includes prerequisites and threats (see Figure 1 of *Article V*) instead of merely listing ecosystem services or individual biodiversity elements as VECs diversifies the planning and goes to the roots of biodiversity loss in the form of pressures caused by land use and transport systems. It helps to diversify treatment of biodiversity issues in planning, moving away from the narrow perspective of dealing with protected species, habitat types, and areas. Furthermore, planners and stakeholders are usually interested more in the benefits that different land-use configurations can provide or preclude than in mere ecological information. This was demonstrated by the great interest of the City of Lahti and the Päijät-Häme regional council in the indicators under the main criteria groups dealing with land use and recreation.

Problems and ways to overcome them

Although it was beneficial to integrate the development work with simultaneous data collection by the researchers and the regional project groups, this work method had its challenges. Because the criteria and indicators' selection was constantly changing, it was difficult for the regional project groups to follow what data had to be collected and in what form. In consequence, data for several indicators had to be collected two or three times and therefore new GIS analyses had to be produced as well. This is inevitable to a certain extent in all development work but could be ameliorated at least in those planning situations in which the final criteria and indicators are used in future. Then it would be possible to give data collectors clear and detailed instructions on what data to collect and how to calculate the indicator values and produce the maps. Planning aids for doing this are to be produced during the latter phases of the project in the form of a detailed Finnish guidance book.

The data problems restricted the choice of criteria and indicators. One can conclude from the experience gained in the project that it is most resource-efficient to use existing widely available databases and indicators that are readily calculated from these rather than try to develop and use indicators for the calculation of which it is impossible to find comparable and consistent data. The future will see data availability issues become less acute with national data improvement projects aimed at standardising national and municipal data collection and increasing the availability of comparable data through data interfaces without at the same time requiring new data collection procedures (SADe 2011). However, these data improvement projects are still in their development phase and do not offer an immediate answer as to how to obtain reliable and comparable ecosystem-services-linked data from municipalities. Therefore, the best solution at present is to use the recently developed criteria and indicator selection in combination with the national data (MSSS and other national databases). However, the analyses of the indicators were rather complicated

and resource-intensive even with the available national data and required advanced GIS skills.

The administrative reality, the very strong boundaries between municipalities, political changes, and relative closeness of planning processes prevented the participatory testing of setting indicator values as targets for ecologically sustainable planning and testing of the criteria in the originally intended milieu: urban regions with land-use challenges that cross municipal borders. The timing of the planning processes did not allow participatory work with stakeholders to test the criteria: in Lahti, the research project fell between the participatory planning phases of the local master planning. Regardless of the success of the co-operation in development of the criteria, opportunities to get involved in the actual planning were not accessible to the researchers. In the Oulu region, the renewal of the master plan was pending for the full three years of the project and ultimately provided opportunities for neither participatory approaches nor researcher involvement in planning. The closest collaboration was for the impact assessment for the Lahti local master plan alternatives within the borders of one municipality. It appears that, although land-use challenges connected with maintenance of biodiversity and ecosystem services call for regional and function-based approaches taking into account human activities across the present administrative borders, the planning system prefers to deal with them within municipalities as long as administrative structures corresponding to the functional structures do not exist. Because most indicator analyses yield data on a 250-by-250-metre grid, the indicators can be used with any spatial delineation. Nevertheless, a very important scoping issue in practical planning is to consider carefully what spatial scales are needed in any given planning situation. The appropriate scale depends on the area subject to planning; the affected area; or the area in which the problem associated with use, maintenance, or threat to a specific ecosystem service is created or can be solved. The solution might be a selection/mixture of indicators with different spatial scales, to describe the most fundamental planning challenges.

7 Discussion

7.1 Knowledge basis in ecological impact assessment and its challenges

My findings demonstrate that the knowledge basis for the comprehensive ecological impact assessment in EIA and municipal land-use planning SEA is far from adequate. Inadequacies persist in the identification and location of the potential environmental stress caused by the project, the area affected, biodiversity elements receiving an impact, the impacts and their prediction, mitigation, and monitoring. The most fundamental shortcomings surround the most elementary issue: what is going to be influenced and how. Surprisingly often impact assessments fail to identify the biodiversity at stake, meaning components, structures, and key ecological processes that are likely to be affected by the project, plan, or programme. In consequence, the selection of the biodiversity elements for baseline studies remains rather haphazard. This is followed by only loose connection between baseline studies and impact prediction. Consequently, impact predictions are vague and not grounded in the collected data, and, therefore, they are difficult to mitigate, not to mention to monitor. My results confirm that the Finnish knowledge base in ecological impact assessment parallels the knowledge base in the other EU member states (Treweek et al. 1997; Thompson et al. 1997; Byron et al. 2000; de Jongh et al. 2004; Gontier et al. 2006) and other parts of the world (Southerland 1995; Warnken and Buckley 1998; Atkinson et al. 2001; Mandelik et al. 2005a; Samarakoon and Rowan 2008; Khera and Kumar 2010). This overall failure of ecological impact assessment to meet the requirements of internationally acknowledged best practices of ecological impact assessment as presented in Subsection 3.3.2 of this work leads one to wonder whether there is something fundamentally erroneous in existing approaches to creating a knowledge basis for biodiversity-inclusive planning and decision-

making? My results point to some factors that may contribute to this failure in the Finnish practice.

Firstly, there is a tendency toward requiring unnecessary detail in relation to certain biodiversity elements, driven by motivation to avoid legal problems associated with strictly protected species and habitat types and at the same time by the need for more broad-brush information for overall biodiversity maintenance. The tendency toward unnecessary detail in SEA has been recognised also by Therivel (2004), Partidário (2007), and João (2007b). Besides avoidance of complaints in the planning, this tendency has been linked to inability to cope with uncertainty of impacts that are indirect and difficult to measure (Noble 2004; João 2007b). This pertains especially to impacts on biodiversity that stem from complex interactions of ecological processes (Erikstad et al. 2007). As noted above, the consultants carrying out ecological baseline studies are reluctant to present something that is not very accurate, because they do not want to risk their reputation (Therivel and Ross 2007). In some cases, very detailed information is needed – for example, in attempts to find out whether a specific species and its habitat on a Natura 2000 site are adversely affected or not – but in many cases some broad-brush land-use data with ecological interpretation would better serve the aim of holistic biodiversity inclusion than detailed mapping of species does.

The most distinctive feature of Finnish ecological impact assessment in EIA, Natura 2000 assessments, and municipal local master planning SEA is the non-existence of land-use data, which is used to some extent in the ecological impact assessment in the UK (e.g., Byron et al. 2000) and, especially, in spatial planning SEA in the Netherlands (Kolhoff and Slootweg 2005). Information on land use does not exist on a detailed level because usually localised data are missing (demonstrated by the infrequent use of maps) and, since no land-use categories are presented, either on a map or in statistics, from the affected area, do not exist even at a broad-brush-level. The only way to cope with absence of data in the impact prediction phase

would be to produce qualitative judgements on biophysical changes in ecosystems in comparison of alternative options in the project or plan by an experienced ecologist without detailed knowledge, as suggested by Slootweg and Kolhoff (2003). This is possible when the ecologist is involved throughout the planning process, but how this could be realised in a local master planning practice wherein ecologists are used only in the baseline study phase while the actual planning and design of alternatives are left to proponents and planners is a major challenge. Another challenge is handling of uncertainty. If impact assessment is viewed solely as a tool for making informed decisions on specific proposals, it is fundamentally unworkable (Wilkins 2003). Information on environmental, economic, and social issues will never be sufficient for predicting effects of a specific project, plan, programme, or policy. The simplification is necessary, and assumptions and uncertainties linked to it should be negotiated more openly and transparently than in present ecological impact assessment practices. The main challenge is to find a balance between broad-brush and detailed information for each planning situation individually.

Secondly, the substantive treatment of biodiversity is not complete. Other aspects of biodiversity than compositional and other levels than species level (and to some extent detailed habitat types) are not considered. In addition, the treatment is usually at absence/presence level. This is very typical in all EIAs in general (Slootweg and Kolhoff 2003); therefore, it did not come as a surprise that the situation in Finland mirrors this quite closely. What was surprising is that my findings revealed that this orientation is predominant in Finnish SEA practices as well. Finnish ecological impact practice in local master planning did not meet the expectations set forth in the biodiversity impact assessment literature for SEA in terms of its handling of ecosystem processes and interactions and its concentration on a broader than single-species perspective (Treweek et al. 2005; Slootweg et al. 2006). My results may suggest that the local master planning is not strategic enough as a planning process fulfilling the po-

tential of SEA. However, this might not be the whole truth. From an earlier SEA study, the quality of impact assessment appears to be even weaker in very strategic-level Finnish SEAs than in EIA (Söderman and Kallio 2009). The same procedural failings appear to characterise all planning types from EIA to more and less strategic SEA. Accordingly, the problem may lay in the substantive orientation. For example, the content of ecological impact assessment in SEA should change from species- and habitat-type-oriented detail-level treatment to broader treatment of environmental characteristics, covering larger areas and ecological processes and their interdependencies. Mapping of individual species and habitat types (or even delineated areas that are of protection value) as a technical exercise without a broad-brush approach dealing with biophysical and social factors and changes – these being on the one hand a prerequisite for maintenance of biodiversity while on the other hand being threats to it – does not offer a usable information base in strategic planning.

Byron et al. (2000) argued that much of the baseline survey effort is wasted because it generates information that contributes little to the prediction for the decision-making, which would require plotting of trends in the status of local populations of species or evaluation of their likely status after the development. The same applies to habitat types and wider ecosystems as well as ecological processes. This argument is still valid. In Finland, the narrow substantive treatment of biodiversity is partly related to the legislation. Both EIA and the land-use and building legislation are more or less procedural legislation not determining the content of the ecological impact assessment; the content comes from sector-based legislation, mainly the Nature Conservation Act (1996). In its present form, the Nature Conservation Act (1996) deals not with broad aspects of biodiversity but only with protection of individual species, narrowly delineated habitat types, and protected areas. The broader biodiversity elements, such as ecological connections, are found only in national land-use guidelines (Valtioneuvoston päätös... 2008). The absence of issues of 1) ecological connectivity between

protected areas and 2) means to prevent fragmentation outside nature conservation areas were also noted by a recent evaluation of Finnish nature conservation legislation (Similä et al. 2010). Similä et al. propose that one way to deal with ecological connectivity is to improve land-use planning practices. A green infrastructure approach (Benedict and McMahon 2006; Opdam et al. 2006; Pauleit et al. 2011) should be made a requirement because of its multifunction and spatial nature and its ability to look at connectivity of biodiversity elements. No matter its type, the legislation – nature conservation legislation, land-use and building legislation, or EIA legislation – should broaden the scope of handling of biodiversity. Greater specificity is possible in legal requirements as to what kinds of biodiversity aspects (component, structure, or key processes) ecological impact assessment would have to address without listing of specific elements (e.g., species and habitat type lists). Terms such as ‘biodiversity’ (used in the EIA legislation) and ‘ecological sustainability’ (used in the land-use and building legislation) are too broad in their definition to give the necessary content to the ecological impact assessment.

Thirdly, Finnish ecological impact assessment practice in EIA or SEA does not take account of the value-laden nature of impact assessment (Beanlands 1988; Richardson 2005); it approaches the information provision task as a technical-rational exercise. However, without very clearly defined requirements concerning what to study and assess impacts on – viz. pre-determined significance determinations in legislation and policies (Slootweg et al. 2006) – the determination of significance is an inevitable part of each assessment and planning situation. Furthermore, it is usually planning-situation-specific, because significance depends on the interests and values of the actors in the planning and impact assessment (Slootweg et al. 2006; Lawrence 2007a). When this is not explicitly acknowledged, the whole assessment exercise is bound to face the problem of inability to select the aspects of biodiversity to address and consequent choice of the easiest way out by taking a superficial approach to ‘everything’ or by concentrating on the most obvious or strictly

protected species, such as the flying squirrel. Therefore, the ecosystem service approach can provide for the highly necessary recognition in Finnish ecological impact assessment practice that what is significant depends on the benefits and ecosystem services provided by biodiversity and on the values that users and beneficiaries assign to these services. Ecological impact assessment and decision-making cannot be separated, with the boundaries between them being blurred (Benson 2003; Wilkins 2003; Richardson 2005; Therivel 2009), and decision-making is always a complex combination of facts and values inseparable from the power of key actors and stakeholders (Owens et al. 2004). Thus both knowledge of ecosystem-service-generating units (Niemelä et al. 2010) for management of biophysical prerequisites and threats to production of certain ecosystem services, as well as stakeholder views as to which ecosystem services ought to be maintained, should form the knowledge basis for ecological impact assessment to be shaped throughout the process. This calls for well-designed composite approaches involving technical, collaborative, and reasoned argumentation (Lawrence 2007a) and building of shared knowledge and learning instead of mere technical information provision (Sheate and Partidário 2010). Accordingly, the challenge in each planning and impact assessment situation is identification of ecosystem services and their users and beneficiaries and of their values, before one can determine which biodiversity elements will be addressed. However, even with its long conceptual history, beginning in the 1990s (Barbier et al. 1994; Costanza et al. 1997; Daily 1997), translating biodiversity to ecosystem services is a rather new approach in Finland (Matero et al. 2003; Naskali et al. 2006) and efforts to open discussion of ecosystem services to wider audiences that include land-use planning and impact assessment practitioners are relatively recent (Saarela and Söderman 2008; Hiedanpää et al. 2010). Clearly, much more effort still is needed to bring ecosystem services as a key significance determinant into day-to-day ecological impact assessment practices of EIA and SEA.

Fourthly, the ecological part of Finnish local master planning SEA appears to lie firmly with the EIA-driven and baseline-oriented SEA school (Partidário 2007), without many objective-led and appraisal-oriented approaches from, for example, political science. It takes the baseline as the most important starting point. Theoretically, a proactive baseline-oriented approach could work if the baseline knowledge were integrated into the planning in such a way that the plan options could utilise the opportunities to enhance biodiversity and the ecosystem services' provision and avoid planning options that cause deterioration in either aspects of biodiversity or the desired ecosystem services (Slootweg et al. 2006). Then a reactive approach in which the main focus is on prediction and mitigation of effects of the plan's alternatives would be unnecessary. However, with the present narrow compositional approach and its lack of knowledge/negotiation related to stakeholder values for ecosystem services, this proactive approach is not feasible and the effects on biodiversity in a broad sense remain largely unpredicted. Therefore, it is alarming that there are planning situations wherein ecological impact assessment is considered to be completed when the baseline studies are completed. The wording of Section 9 of the Land Use and Building Act (1999) also emphasises baselines strongly, by stating primarily that 'plans must be founded on sufficient studies and reports'. Finnish ecological impact assessment in EIA and SEA appears to be parallel to the actual planning process, in contrast to the integrated, environmental-assessment-led or decision-centred models that have been recognised as preferable (e.g., Slootweg et al. 2006; Partidário 2007). What can be gained from political science, SEAs, and EIA theory-building (e.g., Wilkins 2003; Lawrence 2007a, 2007b, 2007c; Wallington et al. 2007; Weston 2010) is undeniable understanding that ecological 'facts' describing biodiversity elements are strongly value-laden irrespective of the approach taken in environmental assessment. Clearly, therefore, it is impossible to produce ecological information that could be taken as a technical baseline as such if the process is external to the planning

and decision-making until this information is contributed at the point in the planning when there is enough of it. Consequently, a merged process of environmental assessment and decision-making appears to be the only feasible way to deal with the value-linked nature of ecological assessment. The challenge is to incorporate this into any planning and impact assessment situation involving biodiversity considerations, as the self-evident starting point, in contrast to the present practice of isolating ecological information production as something ostensibly easily manageable that is to be handled in the first (or last) phases of the planning.

7.2 Restructuring of the ecological impact assessment process and its challenges

My results show that, in addition to exhibiting severe substantive shortcomings, ecological impact assessment fails to meet the procedural requirements set in the European and Finnish EIA and SEA legislation. The procedural failure to meet the legislative requirements is seen especially in the areas of the cumulative impact assessment, provision of baseline information, impact prediction, and monitoring. Neither do the practices fulfil the requirements of internationally acknowledged best practice in ecological impact assessment presented in chapter 3.3.2 of this work. There appears to be a wide gap between 'practices' and 'best practices'. Again, procedural failings are widely reported internationally as well (Southerland 1995; Warnken and Buckley 1998; Treweek et al. 1997; Thompson et al. 1997; Byron et al. 2000; Atkinson et al. 2001; de Jongh et al. 2004; Mandelik et al. 2005a; Gontier et al. 2006; Samarakoon and Rowan 2008; Khera and Kumar 2010). This gap is, in part, inescapable. The best practice established by international environmental policy and by EIA and SEA scholars is followed in EIA and SEA practice with a time lag of several years. To some extent, it is even unfair to compare practices to best practice established years later. It would have been impossible for the ARs of my first study to follow best practices for addressing ecosystem services in

environmental assessment that were not widely recognised internationally until 2002 (CBD 2002; MA 2003). However, many of the procedural best practices originate from the early 1990s (Kennedy and Ross 1992; Morris 1995) and, while still considered valid (Slootweg et al. 2010), are not followed even today.

Accordingly, it is high time for restructuring of the practices of ecological impact assessment. Some phases of the ecological impact assessment process need more emphasis or different handling if they are to be able to contribute broad and comprehensive consideration of biodiversity to planning.

The most severe procedural shortcomings have to do with the inadequate scoping phase. This problem is also recognised internationally (Gray and Edward-Jones 1999; Mandelik et al. 2005a, 2005b). Setting of the boundaries for the impact assessment appears to be neglected: inability to identify the affected areas, select biodiversity elements for further studies in this area, assess the potential spatial and temporal extent of the impacts, and select or even explore methods to study the baseline or impact predictions prevail. While I have not studied the scoping reports of EIA apart from the one of the case study and participation and assessment schemes of local master planning SEA, my analyses imply that the scoping does not meet its legal and best practice requirements. The EIA Act (1999, 2006) requires the assessment programme to present the methods to be used in baseline studies and impact predictions. However, when the final documents fail to present any methods according to which the conclusions on impacts have been drawn, it is fairly clear that the baseline studies and impact prediction were not planned in the first place. As for the best practices for ecological scoping that are listed in chapter 3.3.2, it appears very unlikely that an assessment programme would fully reflect them by stating that ‘the baseline studies will be based on the present information and field checks’ and explaining a bit more about inventories of one species. Neither is stating that the impacts on certain biodiversity elements will be described a sufficient scoping of impact prediction.

The guidelines of the Ministry of the Environment on impact assessment for spatial planning (Paldanius et al. 2006) emphasise the importance of focused impact assessment in view of the impossibility of covering everything. The guidelines assign this task to the participation and assessment scheme and set as a minimum requirement that the scheme present how the impact assessment is linked to the planning process and to what impacts attention will be paid. The guidelines continue by stating that ‘depending on the situation, other issues can be raised, such as alternative options and methods of assessment, baseline studies, and impact predictions’ (Paldanius et al. 2006). My findings imply that even the first requirement is not met, because the consultants often obtain *pro forma* lists to cover everything. The scoping decision on which potential impacts might be the most significant has not been made. The recommendation of raising the other issues is not followed either. In present scoping practices, the central questions – what are the likely impacts of the project or plan, where will they probably occur, and to what biodiversity elements will they probably be directed? – remain continuously unasked. Thus the questions of what to study, how to study it, and how to use the results of the studies to produce meaningful predictions likewise go unasked as well. The universal problem in environmental impact assessment raised by Wathern (1988) of jumping into impact assessment tasks without clearly defined objectives is still as relevant as it was more than 20 years ago. As a consequence of inadequate scoping, the prediction value of baseline studies is low in EIA, Natura 2000 assessments, and local master planning SEA. Therefore, the mitigation and monitoring are not successful either. This is a widely recognised problem in ecological impact assessment (e.g., Treweek 1999). The best practice for scoping (e.g., Bagri et al. 1998; Treweek 1999; Slootweg et al. 2006) – viz. understanding scoping as a kind of preliminary check or ‘mini assessment’ covering all phases and their substantive content from problem definition to monitoring and knowledge requirements in all of these phases – appears to have been misunderstood. It has been reduced to a minimal exer-

cise of giving a consultant a standard blueprint list of biodiversity elements that disregards the specific circumstances of the assessment, such as the biophysical features of the affected area and the knowledge requirements of the planning process and decision-making.

The restructuring that is needed in impact-related practices is a much stronger emphasis on scoping. 'Well begun is half done' applies strongly to ecological impact assessment. In general, more emphasis should be given to the scoping phase and to starting it from a clean slate. Scoping should start with discussion held with stakeholders in the relevant project or plan area and the area likely to be affected. That discussion should explore important questions related to what ecosystem services are needed in view of stakeholders' needs, values, and objectives for the use of biodiversity and ecosystem processes. Discussion is also necessary for understanding which ecosystem functions (composition, structure, and key processes of biodiversity) and biodiversity elements are providing these services and how they are spatially and temporally distributed and linked.

The proponents of the projects and land-use planners should devote much more time and expertise to scoping activities. The authorities should supply guidance and demand more strongly that thorough scoping be performed, including choices regarding the delineation of study and impact areas, the biodiversity elements included and excluded, and the level of detail of assessment needed for different biodiversity elements. The scoping choices should be considered, discussed, decided upon, reasoned on the basis of the views of the users and beneficiaries of the ecosystem services, and reported in the scoping documents. An evaluation of the performance of Finnish EIA legislation listed scoping among the most important development needs for EIA practice (Jantunen and Hokkanen 2010); Jantunen and Hokkanen state that the identification of the most significant environmental impacts should be strengthened and that proponents and authorities should show their reasoning for focusing the studies on them.

My findings indicate that cumulative impacts are usually not addressed at all in Finnish ecological impact practices. The only type involving some cumulative impact assessment is Natura 2000 assessment. In the UK as well, the best treatment of cumulative impact assessment has been in Natura 2000 impact assessments (Cooper and Sheate 2002), but there are still inconsistencies as to what factors are considered cumulative or 'in combination' effects (Therivel 2009) and also Natura 2000 impact assessment reports have less transparency than EIA and SEA do (Uithoven 2010). In Finland, this is emphasised by the fact that there is no public participation in the process, so the process cannot include any joint exercise for significance determination in its scoping. Furthermore, AA reports are not publicly collected so, therefore, are not readily available. This makes it especially difficult to share assessment responsibility across different levels of spatial planning. In theory, the impacts should be assessed at each level of planning with such extent and detail that significant adverse effects are guaranteed not to occur. When the planning becomes more detailed, it should be checked again in the lower tiers of planning that the detailed plans follow the same principle. Then the whole Natura 2000 site and its overall integrity, including its ecological structures and processes, should be treated as a valuable ecosystem component in, for example, regional planning. In the detailed planning, the VECs might be individual habitat types and/or the whole area, depending on how detailed the planning itself is. However, this requires a linkage, tiering, between the planning at regional and municipal level in order to find a reasonable division of assessment tasks. This applies also for distribution of Natura 2000 assessment responsibilities between plans and projects. At the plan level, the AA cannot be final and ascertain that in all circumstances adverse effects are impossible, because the final impacts may depend on the detailed design of the project. However, each plan should ascertain that it enables planning designs that do not cause significant adverse effects. Scott Wilson et al. (2006) recommend so-called 'red flagging', meaning that Natura

2000 sites that could experience adverse effects are flagged first and then revisited later in the plan-making process. If suitable plan designs are not to be found, the red flag remains and the plan needs to be changed. No red flags can be passed on from a higher level of planning to a lower one. Therefore, there must be enough knowledge at the higher planning level with respect to what is feasible or infeasible at the lower planning level.

The restructuring needed for enabling cumulative impact assessment in EIA and SEA and for improving that in Natura 2000 assessment should be a living and iterative linkage between project and plan/programme level in identifying the cumulative impacts suggested by Gunn and Noble (2011). In cases of project types where a programmatic level of sector-scale planning is absent, such as power-line planning, the level of the collaboration should be regional land-use planning. However, tiering mechanisms must exist both on project and at plan/programme level to determine which joint VECs need cumulative effects attention. At the moment, some actors in ecological impact assessment hold the view that certain biodiversity considerations, such as ecological connectivity of areas, are issues of regional planning rather than municipal ones. This would make it impossible to manage the whole multi-level green infrastructure as well as the biodiversity and ecosystems services that are spatially bound to it and dependent on each other at different spatial levels. Usually there is not just one suitable scale for handling a certain issue – there are, instead, multiple or a range of scales needed to identify, assess, and solve planning problems (João 2002, 2007a, 2007b). Without acceptance of the multi-scale planning setting, scale abuse, whether intentional or accidental, is possible. First, scale abuse can take the form of choosing the planning scale such that the assessment leads to the preferred answer rather than emphasising the most significant impacts or solving the problem (João 2007b). Second, assessment questions can be chosen or the problems framed such that they match data that can be collected easily and at little expense (João 2007b). My results show the latter to be the most prevalent form

of scale abuse in ecological impact assessment today. Third, the assessment scale may be defined on the basis of data issues: availability of existing and to-be-collected data (João 2007b). According to my results, availability and non-comparability of municipal data tend to push assessments to remain within administrative borders, hindering treatment of biodiversity issues on the scale at which the problems are caused or could be solved.

It is not very beneficial for biodiversity management to require each individual project or plan to assess just ‘some’ cumulative effects. Important VECs need to be agreed upon through inter-authority and inter-municipal co-operation that crosses planning levels, and each planning level should bear its responsibility for them. Experience has shown that there is too much work and responsibility for each individual project and plan to find all possible information on projects, plans, and other activities in the region that might affect the biodiversity elements addressed in an individual project or plan (Therivel and Ross 2007). My findings confirmed that, in consequence, the cumulative impact assessment remained undone or was seen as completely someone else’s responsibility and cost burden. To work efficiently in, for example, the case of Natura 2000 assessment, this trickle-down effect would mean that also in regional plans, the overall effects of cross-cutting activities, such as urbanisation or urban sprawl, on all Natura 2000 sites should be explored already in Natura 2000 appropriate assessment of region-level plans without details of species and habitats being gone into to a high level of precision. This has been the approach of recent appropriate assessment practice in the UK (Venn and Treweek 2007; Therivel 2009).

There has been discussion of making it obligatory to utilise the results of the EIA similarly to how the results of appropriate assessment are used in permission decisions (Pölonen 2006; Pölonen et al. 2011). Two challenges are linked to shifting the structure of the ecological assessment process towards being more binding in decision-making. Firstly, my results imply that there is not enough substance in ecological impact assessment of EIA and local master plan-

ning SEA to make its application obligatory, on account of the vagueness or non-existence of impact predictions. Secondly, the general-level ecological impact assessment applied in EIA and SEA differs from the last level (project or local detailed planning) in Natura 2000 assessment, which is very detailed and whose judging of impact significance is largely pre-determined by conservation objectives listed in so-called Natura forms according to habitat type and the species annexes of the Habitats Directive (1992, annexes I and II). In terms of the Land Use and Building Act's broad objective, the promotion of ecologically sustainable development and ecological sustainability of land use on the local master plan level (Land Use and Building Act 1999, sections 1 and 39), the scope of ecological impact assessment can be interpreted to be very broad. The same is true for impacts on flora, fauna, and biodiversity as set forth in the EIA Act (1994, 1999, 2006, Section 2). The key difficulty in making it obligatory to apply the results of the ecological impact assessment in EIA and local master planning would be that it would require very specific lists of biodiversity elements for which promotion of ecological sustainability or not causing significant adverse effects is obligatory. These are specific to the environment affected by each individual project or plan and cannot be listed in terms of habitat types and species as for each individual Natura 2000 site. Alternatively, obligation would narrow the focus of ecological impact assessment permanently to the traditional approach of addressing certain protected or designated species, habitat types, and areas without wider focus on all aspects and levels of biodiversity and without any attempt to enhance biodiversity.

However, in addition to more specific legal requirements as to the kinds of biodiversity aspects (again, component, structure, and key processes) that ecological impact assessment would have to address, legal requirements could be added that mandate a certain way of presenting them in the documents and specify added presentational value and influence for the assessment. My results related to local master planning imply that the most important and

most used linkage between a baseline study and a plan design was a map representing the most important results for the baseline in a GIS format that can be directly utilised in planning. At present, ecological impact assessment practice has not realised the potential of communicative use of maps. Jalava et al. (2010) also emphasise presentation issues as among the main elements in need of improvement in Finnish EIA practice. Both description of broader biodiversity aspects and ecosystem services being mandated with new legal requirements and description of these on maps in the scoping, baseline, and impact prediction phase might direct the ecological impact assessment towards a more holistic approach, one that also includes handling of indirect and cumulative impacts.

In summary, to increase the potential influence of ecological impact assessment in planning, procedural features of ecological impact assessment in the form of proper scoping, baseline studies, and prediction need improvement if they are to cover those full procedural steps in actuality and address their substantive content as defined in recent international best practice. In addition, the legislation should be more specific as to the substantive and procedural content of these phases. Methodological choices in particular should be required to be present in such detail that all parties involved in the assessment and planning would be able to understand the decisions concerning what is deemed important to address, at which level, and how the assessment responsibility and work-sharing are to be assigned among the various actors. Also, the relationship between EIA and land-use planning SEA needs legal and practical restructuring in the form of clarification of division of assessment responsibilities between them (Haapanala 2010).

7.3 Collaboration of actors and its challenges

My findings demonstrate that the core actors of the ecological impact assessment – proponents and planners, authorities, and main consultants and sub-consultants – perceive their roles and collaboration needs very differently. The propo-

nents and planners appear to represent the demand side, with a clear assignment to be given to the consultant, representing the supply side. However, the assignment is often too vague, too narrow, or too broad for an ecology consultant to begin the work without scoping. This leaves the actual scoping decisions outsourced to an individual ecologist instead of being considered them as a part of the project or plan preparation process. This parallels the results of Wood et al. (2006) from the UK, where EIA scoping is seen as a technically oriented activity in which consultants take the lead. In Finland, the authorities saw the advantages of collaboration, but their willingness to collaborate and both formal and informal negotiations do not appear to have moved the process toward more collaborative approaches. Ecological impact assessment remains a technical task wherein consultants are expected to provide the information that enables avoidance of significant negative effects on biodiversity. However, the ecology consultants called for closer co-operation with developers, planners, and co-ordinating and guiding authorities. This suggests that, in particular, individual ecology consultants hired to deal with the biodiversity issues are left too isolated to consider scoping questions and value-bound significance determinations related to, for example, the area of impact, selection of VECs, and tiering of green infrastructure issues between planning levels. The challenge is to make all core actors in the planning and impact assessment process, alongside stakeholders, take responsibility for these questions.

Considering ecological impact assessment to be a technical assessment task that is easy to outsource or merely a baseline study task of finding areas that can be set aside before planning continues is rooted in the actors' concepts of biodiversity. It appears that most developers, planners, and consultants use traditional concepts of biodiversity as including only compositional diversity on species level and sometime on a higher level of habitat types and ecosystems. It is the authorities who have the most comprehensive concept of biodiversity. This is in line with the actor views on biodiversity found by Wegner et al. (2005).

However, the concept of ecosystem services is still new to most actors in the field of land-use planning and environmental assessment. According to an interview study by Yli-Pelkonen (2010, in Niemelä et al. 2010), the conceptual term 'ecosystem services' is still unfamiliar to a third of regional environmental authorities. The challenge in enhancing co-operation is to expand the concept of biodiversity beyond narrow boundaries and toward a broad concept that includes uses of biodiversity in the form of ecosystem services. My results point to there being an impetus for all actors to take a more holistic approach to biodiversity considerations. The challenge is to operationalise this need beyond the merely conceptual, abstract understanding that conservation and sustainable use of biodiversity is a holistic, value-linked, and spatially bound matter demanding collaboration of several actors.

My results indicate that regional environment centres monitored the quality of the Natura 2000 assessment process creditably. Some shortcomings in quality supervision by EIA authorities, including disregard for cumulative effects, were found in the EIA process. While general conclusions cannot be drawn from one EIA case., a parallel can be seen here with the Swedish finding that even authorities regard cumulative impact assessment as, more or less, optional in SEA and EIA, not required by EU or national legislation (Wärnbäck and Hilding-Rydevik 2009). Jalava et al. (2010) reported that, regardless of shortcomings in EIA, the overall quality of Finnish ARs is considered rather good both by EIA authorities and by environmental consultants – though, again, this depends on values and perceptions related to what is considered good quality. Without knowing the criteria against which the actors evaluated quality, it is impossible to determine what good quality included in terms of biodiversity. In view of my criteria derived from legislation and the environmental assessment literature, the quality of the parts of EIA that deal with biodiversity issues is low. This demonstrates the need to study in more depth whether EIA authorities' and consultants' perceptions correspond to the broad biodiversity considerations

of the Convention on Biological Diversity and its objectives of assessment of biodiversity impacts through EIA and SEA (CBD 1992, Article 14).

Authorities have a role in supervising EIA, Natura 2000 assessments, and local master planning SEA. Jalava et al. (2010) found that consultants considered the quality of ARs better than authorities did. I found that consultants considered the impacts less negative than did authorities, and also that the planners had a more positive impression of the use of the ecological information than authorities did. In the appropriate assessment, where the role of authorities is the strongest, they are able to shape the project or plan. In EIA, the EIA authority has an important role in guiding the impact assessment. The same holds for authorities, ELY Centres, in local master planning. The guiding role of the authorities in the scoping phase should be strengthened. Firstly, the authorities' skills and knowledge related to biodiversity should be updated, to ensure a broader view of biodiversity and its associated ecosystem services, via training and updated guidance. Secondly, resources and time should be reserved in ELY Centres for guiding and supervising ecological impact assessment processes and ensuring that they are linked to actual planning processes. Thirdly, the planning processes led by planners and developers should reserve enough time for interaction with the authorities. Fourthly, the requirements for the documentation produced in the scoping, including baseline studies' assessment, should be detailed enough to enable quality control. This may demand stricter and more detailed requirements than in the current legislation, as well as further guidelines (Söderman 2003; Paldanius et al. 2006), because the present legal requirements have not had the effect of making scoping choices transparent. However, when documents have a dual role, serving first as sources of information on substantive aspects of the planning and assessment and, secondly, supplying information that is required if one is to be able to evaluate the reliability of the substantive information provided, the requirement of a methodology annex to scoping and assess-

ment reports produced during the planning and assessment could be one solution.

True knowledge brokerage in the form of engaging and building capacity between actors in biodiversity assessment is far from the reality of Finnish ecological impact assessment practices. If ecological impact assessment were to be a real knowledge-sharing exercise, what would be needed? Who should be the broker? Analysis of EIA frameworks in Sweden in road and infrastructure planning from 1995 to 2002 (Isaksson et al. 2009) showed a gradual shift from a 'traditional' planning model emphasising expert knowledge toward a more participatory and dialogue-based model with an expectation that experts should act as dialogue mediators and facilitators in environmental assessment processes that blend and juxtapose different and possibly incompatible logics for how to integrate and resolve competing views, perspectives, and values of actors and stakeholders.

Isaksson et al. (2009) ask whether it is possible, or even desirable, for an expert to shed his or her personal values and views born of professional experience so as to act in a truly unbiased manner in taking such a knowledge-broker role. They argue that it is not. I agree with them. The expert, usually a consultant, would have to act in very different roles in different phases of the assessment, and there would still be a need for ecological (or some other) expertise. In ecological impact assessment, there is a need for biodiversity experts to determine the biodiversity elements, in the form of composition, structure, and key processes, and position them spatially from the perspective of green infrastructure. In addition, there is a need for ecosystem services facilitators / knowledge brokers to link biodiversity elements with users and beneficiaries of ecosystem services derived from those biodiversity elements and with the values and priorities that these parties attach to ecosystem services and to mediate between stakeholders and core actors. Mediation is needed for micro-level informal and key formal decisions throughout the process and for the final formal decision in the form of acceptance of a plan or granting of a permit. Therefore, the knowledge broker should be an 'outsider', a facilitator without a

substantial interest or role in data production for decision-making. Nevertheless, understanding of the link between spatial and temporal aspects of biodiversity and its linkage to ecosystem services is crucial in this knowledge brokerage. Such a knowledge broker's role or actors having that role are absent from present ecological impact assessment practices. Supervising and guiding authorities could act in this role, but this would require a far broader understanding of ecosystem services as well as a more active and stronger role in stakeholder interaction. Facilitating knowledge brokerage could also provide tasks to other types of actors, including consultants with skills in collaborative planning or universities or research institutes with an understanding of biodiversity and ecosystem services in practical planning.

7.4 Promotion of ecological sustainability in environmental assessment and its challenges

Despite the wide array of tools available (Therivel and Wood 2005; Gontier et al. 2006; Bjarnadóttir 2008), my analyses indicate that use of methods and tools more sophisticated than expert judgements and assessment matrices is almost non-existent in Finnish ecological impact assessment practices. GIS methods are used rarely, and when they are used, they are usually applied just to present the project or plan area; study area; or location of protected species, habitats, and sites. Setting thresholds in order to give substantive goals to the impact assessment (Lawrence 2007b), address cumulative impacts and the collaboration needed for their handling (Therivel and Ross 2007; Canter and Ross 2010), and ensure remedial actions in monitoring as appropriate (Trewick et al. 2004) are used extremely rarely in Finnish ecological impact assessment practice. When used, they address only one aspect of biodiversity and usually only one species. The practice displays a planning environment that lacks the time, resources, or skills to carry out impact assessment utilising more than very simple EIA and SEA tools. Even the current guidance for impact assessment in planning (Paldanius et al. 2006)

recommends the use of simple tools and existing data, mentioning as an example the existing spatial datasets and data analysis. Lee (2006) calls for more effective use of simpler assessment methods, use mainly of existing data, and selective use of more complex methods that may require significant quantities of new data. Therefore, a realistic target for improving the basis in knowledge would be effective use of relatively simple tools that utilise existing data as much as possible.

The ecosystem services criteria and indicators were developed in mindfulness of the real-world capacity of EIA and spatial planning SEA to utilise mainly simple tools, techniques, and approaches. For several reasons, I consider it important to use separate ecological criteria instead of general, integrated sustainable development criteria. The first reason is the documented and criticised tendency of the triple-line approach to sustainability in EIA and SEA to undermine environmental considerations as compared to economic ones (Kørnøv and Thissen 2000; Morrison-Saunders and Fischer 2006; Hilding-Rydevik and Bjarnadóttir 2007; Kidd and Fischer 2007). The second reason is the necessity of retaining the substantive clarity of assessment tasks, keeping environmental arguments separate from socio-economic ones (Therivel 2004; Wallington et al. 2007). The intention is to avoid an 'SD smoothie' – a mix wherein everything is over-integrated and the concept of sustainability is made unappealing and difficult to distinguish from its constituent ingredients (Morrison-Saunders and Fischer 2006). The third reason is the role of trade-offs. Sustainability is about not balancing but multiple reinforcing gains. Therefore, trade-offs/sacrifices are acceptable only as a last resort, when all other options have been found to be worse (Gibson 2005). They are acceptable only when maximum net gains are delivered, when the trade-offs are proved to avoid significant adverse effects on sustainability, and when these are openly discussed in stakeholder involvement (Gibson 2005). It follows that there are factors that cannot be traded away, and ultimately the sustainability choice evaluated by means of all three pillar criteria will be rather

narrow. This must be acknowledged in practical planning: not everything is possible, even if the planning choices are sustainable.

Given that the substantive orientation of EIA and of SEA were primarily environmental and they had an environmental advocacy role, the role of EIA and SEA should be to ensure that planning choices are environmentally sustainable. In the case of triple-line sustainability appraisal (George 2001), the pillars should be treated as equal. To avoid a mixture wherein one cannot distinguish what is, for example, environmentally or economically sound, treating the pillars separately would still be beneficial before comparison and integration. Morrison-Saunders and Fischer (2006) call equal treatment of pillars 'genuine' sustainability assessment, in contrast to that dominated by economic priorities. Accordingly, a sustainable solution would be only one that is both ecologically viable and socially and economically acceptable (Potschin and Haines-Young 2006). However, it is not so straightforward a task to separate bio-physical, non-human aspects of impact assessment. Therefore, ecosystem services represent something that is strongly bio-physically bound to green infrastructure but inevitably examined through a lens of human socio-economic valuations. In discussion of benefits and losses in different land-use scenarios, ecosystem services and their value-laden nature cannot be excluded when there is stakeholder involvement (de Groot et al. 2006; Pauleit et al. 2011). The ecosystem services criteria then are a kind of 'mix and match' combination with an environmental focus.

It is impossible to specify significance determinations in legislation and environmental policies regarding bio-physical features of the environment such that these are easily translatable to the biodiversity elements for merely baseline-oriented assessment. Firstly, these features are site- or region-specific. Secondly, they are planning-situation- and stakeholder-specific. Interpretation of ecological sustainability needs to be planning-situation specific, on account of the different values and expectations attached to sustainable development in planning settings by planners, politicians, and

stakeholders. Therefore, a mixture of baseline-led and objective-led approaches is needed. Setting of targets or threshold levels for attainable or acceptable development is a precondition for an objective-led approach. Therefore, it is necessary to set sustainability criteria or thresholds that cannot not be crossed (Sadler 1999; George 2001; Noble 2002; Pope et al. 2004; Gibson 2001, 2005; Opdam et al. 2006). This is a prerequisite for ability to identify and manage cumulative impacts in a regional setting (Gunn and Noble 2009). Gunn and Noble argue (2009) that regional SEA should ultimately place less emphasis on predicting impacts with great precision and put more emphasis on setting targets for regional environmental protection and development. In SEA processes, ecosystem services thinking has resulted in transparency of planning and facilitated sustainability by recognising economic, social, and environmental benefits and development needs and has identified winners and losers from certain changes (Slootweg and van Beukering 2008). The ecosystem services indicators help planning to do exactly that. Compared to earlier, merely conceptual criteria (Pope et al. 2004), whose sustainability aspects were very much open to interpretation, the spatial indicators concretise the abstract thresholds to a level at which they can be discussed among core actors of impact assessment and stakeholders. Further, winners and losers are identifiable because the impacts of changes can be localised and demonstrated on a map. To avoid a technical-rational approach, certain openness to interpretation is needed, but at the same time something concrete – preferably quantitative – should be available to enable the target-setting, impact assessment, and follow-up to ensure that the targets are met and thresholds are not crossed. Qualitative second-order criteria serve the purpose of openness, and quantitative indicators serve the purpose of concreteness. It is still important to realise that the whole of planning and assessment is very much qualitative. The quantitative thresholds should not be used for 'hiding' behind one single indicator value without looking holistically at ecological sustainability (Wood 2008). Furthermore, neither absolute indicator thresholds

nor the spatial scales on which the indicator values are calculated should be manipulated during the planning in order to best match the (often primarily economically set) development goals (Karstens 2007; Wood 2008).

Both baseline- and target-indicator-oriented approaches are always to some extent data-driven. Nevertheless, I would argue that the assessment is not scale-abusive (João 2007b) when it is using the ecosystem services criteria and indicators, because the criteria and initial indicators were developed firstly on the basis of conceptual understanding of ecological sustainability and ecosystem services and also from best practices of biodiversity-inclusive planning and after that the indicators for which there were inconsistent data were eliminated. Therefore, the indicators are as good as they can be in view of currently available spatial data. More numerous and versatile indicators could have been developed on the basis of statistics available on the municipal level but without 'spatial thinking'. However, this would not have brought anything new to environmental assessment practices. Indicators telling more about biodiversity, especially the ecosystems at stake, could have been developed on the basis of, for example, endangered habitat types (Raunio et al. 2008), but such indicators would have been impossible to realise in practical planning with predominantly the existing data. In the absence of an existing information basis, ecological impact assessment with such an indicator would turn into a very laborious exercises of mapping those habitat types and neglecting the other aspects of biodiversity. However, as more data become available consistently throughout the country, the range of indicators can be broadened. It also must be remembered that, for ability to handle multidimensional and complex issues, considerable simplification is needed.

Regardless of the constraints linked to the use of GIS and spatial data, the three pilot cases involved in the development and testing of the criteria experienced the tool as very useful. However, the final test of the usability of the criteria and indicators is the planning processes, which will use them independently, without external assistance from researchers. Constraints

paralleling Finnish experiences of using spatial data in environmental assessment processes have been reported from Ireland (González et al. 2011). The Irish starting point for development of a GIS-data-based approach to impact assessment was Annex 1 of the SEA Directive (CEC 2001), describing the content of environmental reports corresponding to Section 4 of the Finnish SEA Decree (2005) and annexes to the INSPIRE Directive (CEC 2007), but the Irish list of the data collected is very similar to the data needed for the ecosystem services indicators. The Irish team used a weighted overlay technique calculating vulnerability scores for the grids, whereas the ecosystem services indicator approach avoided all weighting, aggregation, and index-type approaches, to avoid confusion and preserve transparency in stakeholders' involvement and forming of opinions on targets. Nevertheless, the data needs were still the same.

González et al. (2011) in their five pilots testing GISSEA methodology faced similar timing constraints for fitting the data delivery schedule to the decisional scale to those in the three Finnish pilots. The collection of data was slow and delayed by several months in both countries, and it was beset with data accessibility and inconsistency constraints, data conversion issues, and data improvement tasks. In the Irish case, the weighted overlay method was excluded in those planning processes with limited time, in favour of more urgent and basic SEA tasks, such as preparation of baseline maps and definition of alternatives (González et al. 2011). This supports my view that the simpler the method is, the more likely it is to be used. However, it was recognised in the Finnish pilots that even though the indicators were simple their calculation was not, and even GIS professionals used to handling the present data spent considerable time on the calculations. Accordingly, the spatial indicators may not work well enough even with the upcoming Finnish user guidelines, unless they are all included as standard analyses in the Finnish monitoring system of spatial structure (MSSS), where they would be readily available as standard material for analysis of 250 x 250 metre grids to be adopted rapidly in

the planning situation and scale at hand. Therefore, all of the indicators should be included as standard analyses in MSSS, so as to be readily available and usable with basic GIS skills. Many more years would have been needed in both the Irish and Finnish pilots to test the functionality of the tool throughout all planning processes from target-setting to monitoring. As it was, testing could occur in only certain phases of the assessment. Furthermore, both the Irish and the Finnish approach failed to involve stakeholders. The Irish pilot workers tried to involve stakeholders through an Internet site but did not succeed, because of technical barriers, stakeholders' preference for giving feedback in written form, and their lack of interest in getting involved in strategic planning where no implications of land use are spatially identifiable yet (González et al. 2011). This lack of public interest in highly strategic impact assessment processes has been reported from Finland as well (Söderman and Kallio 2009). The Finnish pilot work sought involvement in participatory planning and assessment situations but without success. The researchers were not acting as knowledge brokers; their role resembled rather more that of ecological consultants.

The future will present challenges to use of the ecosystem services criteria and indicators in practical planning and environmental assessment. The first of these involves their independent use by planners in different planning processes. The criteria and indicators are unavoidably supply-driven, as they were mainly developed by researchers – even though in close collaboration with the planners of the pilots. Furthermore, the criteria and indicators are not definitively 'value-free'; they were affected by the values of all researchers and planners involved, including my own. Each planning situation needs to be very transparent and methodologically open with respect to how the threshold values are set: on what knowledge they are based, and by whom and in what target-setting process they are set. The same requirement for methodological transparency applies to impact assessment. After dissemination of the criteria and indicators and also of the methodology guidelines to users, further

research is required, to explore how the criteria and indicator values are used in independent planning processes without support: are they used as technical facts or mediating planning aids that enhance discussion and target-setting as they were originally meant to do? Are they usable with basic GIS skills or too elaborate to use?

The second challenge involves the scaling. Work should be undertaken to explore further how spatial levels and dimensions of sustainability are treated in planning styles wherein each planning level is responsible for dealing with relevant questions and thresholds of that level and these were then trickled down or evaporated up to other tiers of planning. In practice, this should work in both directions (Arts et al. 2005; Gunn and Noble 2011), enabling use of the criteria at all levels. In addition, the use of ecological criteria side by side with social and economic criteria should make the real or 'genuine' sustainability choice space transparent. It might be much narrower than had been thought, when one takes seriously the thresholds inherited from higher tiers of planning – for example, the national guidelines on land use (Valtioneuvoston päätös... 2008). The spatial indicators enable a spatial dimension to the goals. This requires that objectives and policies be formulated in spatial terms (González et al. 2011). Consequently, the planning system and its actors would be committed to following the spatial thresholds set. For example, spatial targets related to the ecological connections established in regional planning would not be available for reopening in municipal land-use planning. Thus, gaining actors' and stakeholders' collaboration and commitment to managing impacts that are mainly indirect and cumulative in nature, such as impacts on biodiversity, is the most pressing challenge. It must be recognised in practical planning that cumulative effects require cumulative mitigation and management solutions (Canter and Ross 2010). In addition, setting the boundaries to the spatial units within which the cumulative issues are to be handled is challenging, as demonstrated in the testing in the Lahti and Oulu urban regions. When administrative boundaries are used, the 'big picture'

is lost. When functional boundaries are used, whether human or ecological, the planning system is unable to follow the boundaries and unable to commit itself. Again, a mixture of spatial boundaries would appear the most efficient approach.

The third challenge involves data. At present, it appears that some data problems associated with spatial data will lessen in their impact at a pace with progress in national data improvement projects, including plans to make available a large quantity of biodiversity-ecosystem-services-related data among other data (SADe 2011). In addition, there is already MSSS (SYKE 2011) in place to store and retrieve data and analyses. The more data and analyses become available, the more important it is to include metadata and explanations describing what can be done with qualitative and quantitative data and stating what said data can or cannot indicate. Especially at strategic levels of assessment, presenting findings in a quantitative form may create an exaggerated and misleading impression of accuracy (Lee 2006). In an ecological impact assessment and planning culture still largely dominated by technical-rational ideals of definite ecological facts steering the planning – although there are many signs of a shift in the planning paradigm towards ecosystem services thinking (Hiedanpää et al. 2010) – it is essential to consider data as something aiding in planning choices rather than offering immediate, ready answers.

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