CHAPTER 1

ASSESSMENT OF ECOSYSTEMS RISKS

Over the last decades direct and indirect environmental effects of human activities has become a focus of special attention of the general public, state authorities, and international organizations. A number of approaches to predict, evaluate, and mitigate human-induced alterations in the biophysical environment have emerged including environmental impact assessment (EIA). EIA has become a powerful tool to prevent and mitigate environmental impacts of proposed economic developments.

In the current EIA practice, impacts on natural systems (ecological effects) are often given less attention than they deserve (Treweek, 1999). One of the key reasons is a great deal of uncertainty associated with ecological impact studies.

Meanwhile, there has arisen a well established methodology for assessing developments in the face of a high degree of uncertainty and establishing the potentially high significance of impacts, we call this methodology *risk assessment* (*RA*) including *environmental risk assessment* (*ERA*). Recent interest in "tools integration" (Sheate, 2002) is related to growing debate on the benefits of integrating RA into EIA procedures in terms of improving treatment of impacts of concern (see, e.g., Andrews, 1990; Arquiaga et al., 1992; NATO/CCMS, 1997; Poborski, 1999). A number of procedural and methodological frameworks for EIA–RA integration has already been proposed and many researchers believe that RA should be used extensively in assessment s for many types of impacts including impacts on ecosystems (Lackey, 1997).

Ecological impact assessment induced by various human activities is a focal point of improving methodology for environmental impact assessment. Although there is an established methodology for assessing EIA, it is applied mainly in an *ad hoc* manner (Eduljee, 1999). Moreover, there is a vocal critique on applicability of ERA methodology to studies of ecosystem effects of proposed development (Lackey, 1997). The state-of-art ecological risk assessment (EcoRA) has established tools and techniques for and provides credible findings at species level investigations. Recent developments in ERA methodology allowed the researcher to move to population and even community level assessments (see Smrchek and Zeeman (1998) for details). At the same time, formal EcoRA is sometimes focused on effects on groups of organisms, and not an ecosystem as a whole. RA at ecosystem level is usually comparative and qualitative (Lohani et al., 1997).

Meanwhile, a quantitative approach to assessing pollution effects on ecosystems has already been developed. A Critical Load and Level (CLL) concept has been used for defining emission reduction strategies under the UNECE Convention on

Long-range Transboundary Air Pollution (LRTAP). Over time, the critical load approach has been defined not only at international but also at regional and local levels (Posch et al., 1993, 1997, 1999, 2001, 2003; Bashkin, 1997, 2002).

Accordingly, this chapter discusses the incorporation of the CLL concept into EIA for assessment and management of risks for natural ecosystems. The authors aimed at providing insights on applying this effect-oriented approach within a legally established procedure for assessing proposed economic developments. The proponents are encouraged to consider the CLL methodology as a promising tool for cost-effective impact assessment and mitigation (Posch et al., 1996).

The first section explains the concepts of EIA and RA and the existing approaches to their integration. This is followed by an analysis of the current situation with ecological input into EIA and discussion on how the formal EcoRA framework provides for site-specific ecosystem risk assessment. The subsequent section reviews the CLL approach and its applicability for assessing ecological effects in EIA. Finally, a model for assessment of ecosystem risks within EIA using the CLL approach is proposed.

1. CONCEPTS OF ENVIRONMENTAL IMPACT ASSESSMENT AND RISK ASSESSMENT AND APPOACHES TO THEIR INTEGRATION

The technique of risk assessment is used in a wide range of professions and academic subjects. Accordingly, in this introductory section some basic definitions are necessary.

Hazard is commonly defined as "the potential to cause harm". A hazard can be defined as "a property or situation that in particular circumstances could lead to harm" (Smith et al., 1988). Risk is a more difficult concept to define. The term risk is used in everyday language to mean "chance of disaster". When used in the process of risk assessment it has specific definitions, the most commonly accepted being "The combination of the probability, or frequency, of occurrence of a defined hazard and the magnitude of the consequences of the occurrence" (Smith et al., 1988).

The distinction between hazard and risk can be made clearer by the use of a simple example. A large number of chemicals have hazardous properties. Acids may be corrosive or irritating to human beings for instance. The same acid is only a risk to human health if humans are exposed to it. The degree of harm caused by the exposure will depend on the specific exposure scenario. If a human only comes into contact with the acid after it has been heavily diluted, the risk of harm will be minimal but the hazardous property of the chemical will remain unchanged.

There has been a gradual move in environmental policy and regulation from hazard-based to risk-based approaches. This is partly due to the recognition that for many environmental issues a level of zero risk is unobtainable or simply not necessary for human and environmental protection and that a certain level of risk in a given scenario is deemed "acceptable" after considering the benefits.

Risk assessment is the procedure in which the risks posed by inherent hazards involved in processes or situations are estimated either quantitatively or qualitatively. In the life cycle of a chemical for instance, risks can arise during manufacture, distribution, in use, or the disposal process. Risk assessment of the chemical involves identification of the inherent hazards at every stage and an estimation of the risks posed by these hazards. Risk is estimated by incorporating a measure of the likelihood of the hazard actually causing harm and a measure of the severity of harm in terms of the consequences to people or the environment.

Risk assessments vary widely in scope and application. Some look at single risks in a range of exposure scenarios such as the IPCS Environmental Health Criteria Document series, others are site-specific and look at the range of risks posed by an installation.

In broad terms risk assessments are carried out to examine the effects of an agent on humans (Health Risk Assessment) and ecosystems (Ecological Risk Assessment). Environmental Risk Assessment (ERA) is the examination of risks resulting from technology that threaten ecosystems, animals and people. It includes human health risk assessments, ecological or ecotoxicological risk assessments, and specific industrial applications of risk assessment that examine end-points in people, biota or ecosystems.

Many organizations are now actively involved in ERA, developing methodologies and techniques to improve this environmental management tool. Such organisations include OECD, WHO and ECETOC. One of the major difficulties concerning the use of risk assessment is the availability of data and the data that are available are often loaded with uncertainty.

The risk assessment may include an evaluation of what the risks mean in practice to those effected. This will depend heavily on how the risk is perceived. Risk perception involves peoples' beliefs, attitudes, judgements and feelings, as well as the wider social or cultural values that people adopt towards hazards and their benefits. The way in which people perceive risk is vital in the process of assessing and managing risk. Risk perception will be a major determinant in whether a risk is deemed to be "acceptable" and whether the risk management measures imposed are seen to resolve the problem.

Risk assessment is carried out to enable a risk management decision to be made. It has been argued that the scientific risk assessment process should be separated from the policy risk management process but it is now widely recognised that this is not possible. The two are intimately linked.

Risk management is the decision-making process through which choices can be made between a range of options that achieve the "required outcome". The "required outcome" may be specified by legislation using environmental standards, may be determined by a formalized risk–cost–benefit analysis or may be determined by another process for instance "industry norms" or "good practice". It should result in risks being reduced to an "acceptable" level within the constraints of the available resources.

Risks can be managed in many ways. They can be eliminated, transferred, retained or reduced. Risk reduction activities reduce the risk to an "acceptable" level, derived after taking into account a selection of factors such as government policy, industry norms, and economic, social and cultural factors.

It is important to note that although risk assessment is used extensively in environmental policy and regulation it is not without controversy. This is also true for risk management.

2. BIOGEOCHEMICAL APPROACHES TO ENVIRONMENTAL RISK ASSESSMENT

It is well known that biogeochemical cycling is a universal feature of the biosphere, which provides its sustainability against anthropogenic loads, including acid forming compounds. Using biogeochemical principles, the concept of *critical loads* (CL) has been firstly developed in order to calculate the deposition levels at which effects of acidifying air pollutants start to occur. A UN/ECE (United Nations/Economic Committee of Europe) working Group on Sulfur and Nitrogen Oxides under Long-Range Transboundary Air Pollution (LRTAP) Convention has defined the critical load on an ecosystem as: "*A quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge*" (Nilsson and Grennfelt, 1988). These critical load values may be also characterized as "*the maximum input of pollutants (sulfur, nitrogen, heavy metals, POPs, etc.), which will not introduce harmful alterations in biogeochemical structure and function of ecosystems in the long-term, i.e., 50–100 years*" (Bashkin, 1999).

The term *critical load* refers only to the deposition of pollutants. Threshold gaseous concentration exposures are termed *critical levels* and are defined as "*concentrations in the atmosphere above which direct adverse effects on receptors such as plants, ecosystems or materials, may occur according to present knowledge*".

Correspondingly, transboundary, regional or local assessments of critical loads are of concern for optimizing abatement strategy for emission of polutants and their transport (Figure 1).

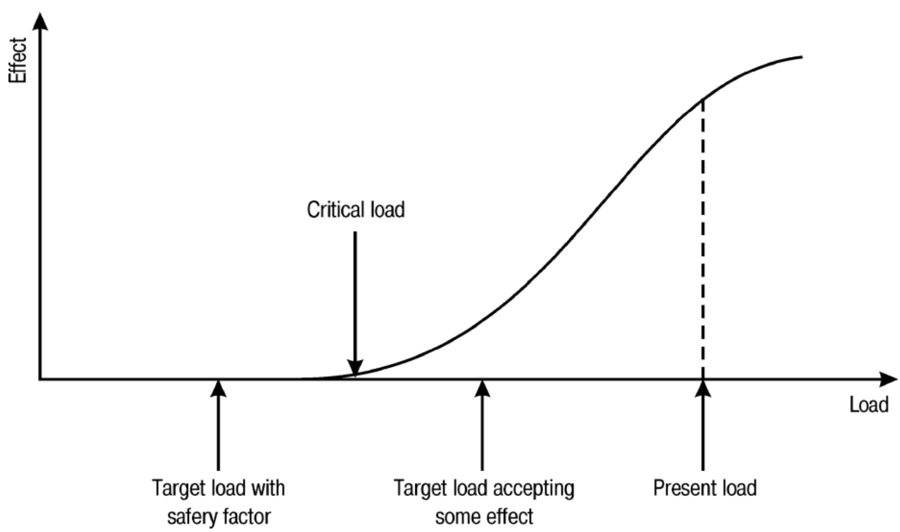

Figure 1. Illustration of critical load and target load concepts.

The critical load concept is intended to achieve the maximum economic benefit from the reduction of pollutant emissions since it takes into account the estimates of differing sensitivity of various ecosystems to acid deposition. Thus, this concept is considered to be an alternative to the more expensive BAT (Best Available Technologies) concept (Posch et al., 1996). Critical load calculations and mapping allow the creation of ecological–economic optimization models with a corresponding assessment of minimum financial investments for achieving maximum environmental protection.

In accordance with the above-mentioned definition, a critical load is an indicator for sustainability of an ecosystem, in that it provides a value for the maximum permissible load of a pollutant at which risk of damage to the biogeochemical cycling and structure of ecosystem is reduced. By measuring or estimating certain links of biogeochemical cycles of sulfur, nitrogen, base cations, heavy metals, various organic species and some other relevant elements, sensitivity both biogeochemical cycling and ecosystem structure as a whole to pollutant inputs can be calculated, and a “critical load of pollutant”, or the level of input, which affects the sustainability of biogeochemical cycling in the ecosystem, can be identified.

3. INTEGRATION OF RISK ASSESSMENT AND ENVIRONMENTAL IMPACT ASSESSMENT FOR IMPROVED TREATMENT OF ECOLOGICAL IMPLICATIONS

EIA is a process of systematic analysis and evaluation of environmental impacts of planned activities and using the results of this analysis in planning, authorizing and implementation of these activities. Incorporation of environmental considerations into project planning and decision-making has become a response to growing public concern of potential environmental implications of economic activities. Over the last decades EIA has become a legally defined environmental management tool implemented in more than 100 countries worldwide (Canter, 1996).

A generic model of the EIA process includes such distinct stages as screening, scoping, impact prediction and evaluation, mitigation, reporting, decision-making, and post-project monitoring and evaluation (EIA follow-up) with public participation and consideration of alternatives potentially incorporated at all stages of the process (Wood, 1995; Canter, 1996; Lee and George, 2000).

A special assessment procedure that aims at tackling uncertain consequences of human activities is called risk assessment (RA). The main objective of risk assessment is to use the best available information and knowledge for identifying hazards, estimating the risks and making recommendations for *risk management* (World Bank, 1997).

Traditionally, RA has been focused on threats to humans posed by industrial pollutants. In recent times there has been a shift to other types of hazards and affected objects (Carpenter, 1996). Ecological risk assessment (EcoRA) has already evolved into separate methodology under the general RA framework.

When applied to a particular site and/or project, RA procedures include several generic steps such as *hazard identification, hazard assessment, risk estimation* and *risk evaluation*.

Often contrasted in conceptual terms, EIA and RA have a common ultimate goal — "the rational reform of policy-making" (Andrews 1990). Both assessment tools are intended to provide reasoned predictions of possible consequences of planned decisions to facilitate wiser choices among the alternatives. To link risk assessment and impact assessment paradigms one can suggest a definition of *environmental impact* as any change in the level of risk undergone by receptors of concern that are reasonably attributable to a proposed project (Demidova, 2002).

The following reasons for integrating EIA and RA are frequently distinguished. On one hand, it has been presumed that EIA can benefit from utilizing RA approaches, in particular in order to improve the treatment of human health issues and uncertain impacts. It has been argued that RA could make impact prediction and evaluation more rigorous and scientifically defendable. Beyond impact analysis, RA can facilitate analysis of alternatives and impact mitigation strategies. Apart from obvious benefit for impact assessors this would provide for "greater clarity and transparency in decision making" (Eduljee, 1999) and help manage risks at the project implementation stage. On the other hand, the integration might help to institutionalize the RA procedure in the framework of such a widely used decision-support tool as EIA. It may also enhance RA with public participation and consultation elements borrowed from EIA.

Few jurisdictions have mandatory legal provisions for RA application within EIA (e.g., Canada, USA (Smrchek and Zeeman, 1998; Byrd and Cothern, 2000)). There is no universally agreed methodological and procedural framework to integrate RA into EIA and only a limited number of practical recommendations for improvements in the EIA process that would facilitate such integration. Nevertheless, many researchers linked comprehensive impact assessment with using "scientifically based" risk assessment methods (see, e.g., Andrews, 1990; Arquiaga et al., 1992; Canter, 1996; Lackey, 1997).

Moreover, a number of approaches for EIA–RA integration have already been proposed (see, e.g., Arquiaga et al., 1992; NATO/CCMS, 1997; Eduljee, 1999; Poborski, 1999). Most of them follow the widely accepted idea of "embedding" risk assessment into EIA and incorporating RA methods and techniques into EIA methodology; they are organized according to the sequence of generic EIA stages discussed above (see Demidova (2002) for in-depth discussion). A general model for integrating RA into EIA, which summarizes many of them, is presented in Demidova and Cherp (2004).

4. ASSESSMENT OF ECOSYSTEM EFFECTS IN EIA: METHODOLOGICAL PROMISES AND CHALLENGES

Any changes in the environment resulting from the proposed projects including impacts on ecosystems are under the EIA scope. At the same time, the traditional focus of EIA is the quality of environmental media: ambient and indoor air, water, soil parameters of human biophysical environment. According to reviews of EIA practice, potential impacts of proposed developments on biota and natural ecosystems has often been assessed superficially and even neglected (see, e.g., Treweek et al., 1993; Treweek, 1995; Treweek, 1996; Thompson et al., 1997; Byron et al., 2000).

Firstly, this situation can be linked with a relatively strong anthropocentric tradition in environmental management and a utilitarian approach to natural resource use. Since scoping of impacts and differentiating among significant and insignificant impacts at an early stage of the assessment process is among key EIA features, an assessor can potentially overlook the importance of ecosystem change, rank these effects as insignificant and not include them in EIA ToR for detailed investigation.

Secondly, internal complexity of natural systems makes prediction of changes in the ecosystem functioning an extremely difficult task. The higher the natural system, the higher the complexity and lower predictability of its response to influence of stressors. Many existing impact prediction methods (including simulation modeling) imply a number of simplifications that generate high uncertainty, which undermines credibility of the findings. In addition, modeling of processes in living systems (from an organism to an ecosystem) requires collecting comprehensive input datasets. It may take a lot of resources to compile such a database (either by desk or field studies). However, the output of this hard work may be of little value due to high data and/or decision uncertainty. Lack of scientific evidence is a key reason to avoid conducting quantitative assessments of ecological impacts and even considering these issues in EIA.

Meanwhile, failure to quantify ecological impacts is among key shortcomings of ecological impact assessment (Treweek, 1999). In current practice quantification usually stops at defining the level of predicted concentration of pollutants in the environmental media and few assessors go further to assess actual effects on biological receptors—organisms, populations, communities, and ecosystems (Arquiaga et al., 1992; Treweek, 1996, 1999). At the same time many projects, especially greenfield developments, are associated with impacts on the natural ecosystems that are of high significance (e.g., if a protected area is to be potentially affected) that requires rigorous ecological impact assessment.

A number of EIA theorists believe in incorporating formal RA methods into EIA as a way to cope with uncertainties, especially in impact prediction where a formal framework for ecological risk assessment (EcoRA) is already developed. It includes three generic phases: *problem formulation*, *analysis*, and *risk characterization* followed by *risk management.* The analysis phase includes an *exposure assessment* and an *ecological effects assessment* (see, e.g., US EPA (1998)).

Despite rapid development of EcoRA guidance and wide support for the idea of tools integration, ecological risk assessment is rather an exclusion in EIA practice. In fact, the formal risk assessment follows the "bottom–up" approach to assessing ecosystem-level effects. The assessor depends mainly on findings of laboratory toxicity testing that are extrapolated to higher levels of natural system hierarchy (from organisms to communities and even ecosystems) using various factors (Smrchek and Zeeman, 1998). Meanwhile, too many assumptions put a burden of high uncertainty on final quantitative risk estimates. Moreover, ecosystem risk assessments of this type are rather experiments than established practice. High costs and lack of required data are among key reasons for avoiding this approach by practitioners.

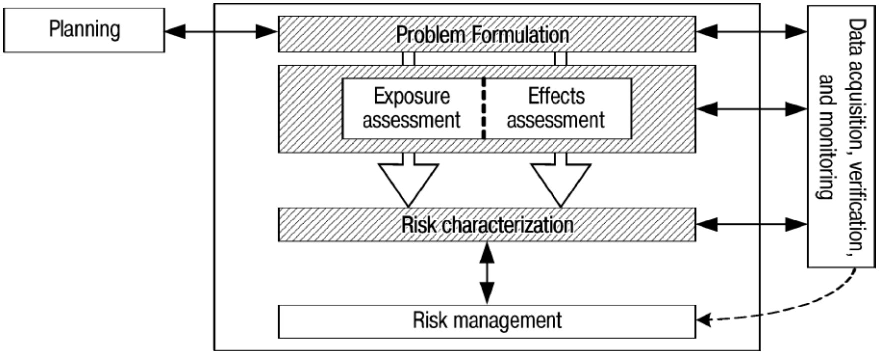

Figure 2. The framework for ecological risk assessment (from U.S. EPA, 1998).

As a result, an EIA practitioner faces considerable difficulties while assessing impacts on ecosystems. On one hand, there are legal requirements to assess fully ecological effects and best practice recommendations to undertake quantitative assessments where possible. On the other hand, many assessors lack tools and techniques to undertake estimations with a high degree of confidence and prove them to be scientifically defensive. Of importance, there are formal RA techniques for tackling the uncertainty[1] (first, data uncertainty) in a clear and explicit manner and its quantification, to increase impact predictability.

As to assessment of ecosystem impacts, the proposed integration model implies using formal EcoRA methodology. The general EcoRA framework suggested by the US Environmental Protection Agency is depicted in Figure 2. It is similar to schemes followed by other counties.

Ecological risk assessment in EIA is to evaluate the probability that adverse ecological effects will occur as a result of exposure to stressors[2] related to a proposed development and the magnitude of these adverse effects (Smrchek and Zeeman, 1998; US EPA, 1998; Demidova, 2002). A lion's share of site-specific EcoRAs were concerned with chemical stressors—industrial chemicals and pesticides.

In formal EcoRA framework three phases of risk analysis are identified: *problem formulation*, *analysis*, and *risk characterization* followed by *risk management.* The analysis phase includes an *exposure assessment* and an *ecological effects assessment* (see Figure 2).

The purpose of *problem formulation* is to define the rationale scope, and feasibility of a planned assessment process. The key implication for EcoRA is a concern that

[1] The two most widely known are sensitivity analysis and Monte Carlo error analysis (see De Jongh (1990) for in-depth discussion).

[2] *Stressor* is a chemical, physical or biological agent that can cause adverse effects in non-human ecological components ranging from organisms, populations, and communities, to ecosystems (Smrchek and Zeeman, 1998).

something is or will be wrong with the environment. In response to this suspected problem, available information on stressors, effects, and receptors is analyzed to select risk assessment endpoints (assessment and measurement endpoints) and possible conceptual models. In addition, policy and regulatory requirements, available budget and an acceptability level of uncertainty are considered to develop a plan for EcoRA (analogous to EIA ToR) to determine which key factors to explore. The latter is a point where risk assessors and managers should interact closely to ensure the success of assessment process and final decision-making (Byrd and Cothern, 2000; Smrchek and Zeeman, 1998).

In the analysis phase, risk assessors examine exposure to selected stressors and resulting effects in receptors (including ecosystems or environmental compartments). An *exposure assessment* aims at identifying and quantifying stressors that are causing the problem by examining physical and chemical measurements and observing biotic indices. The *ecological effects assessment* links the degree of exposure (e.g., concentrations of contaminants in exposure media) to adverse changes in the state of receptors. First, data on effects of a stressor are categorized using toxicity testing known as the "dose–response" curve. Second, the evidence is weighted if the identified hazard is of practical significance (Smrchek and Zeeman, 1998).

In the final phase of risk analysis—*risk characterization*—one integrates outputs of effects and exposure assessments. Risk is expressed in qualitative or quantitative estimates by comparison with reference values (e.g., hazard quotient). The severity of potential or actual damage should be characterized with the degree of uncertainty of risk estimates. Assumptions, data uncertainties and limitations of analyses are to be described clearly and reflected in the conclusions. The final product is a report that communicates to the affected and interested parties the analysis findings (Byrd and Cothern, 2000).

Risk characterization provides a basis for discussions of *risk management* between risk assessors and risk managers (US EPA 1998). These discussions are held to ensure that results of risk analysis are presented completely and clearly for decision makers, thus allowing any necessary mitigation measures (e.g., monitoring, collecting additional data to reduce uncertainty, etc.).

At present conducting EcoRA is rather an exclusion in EIA practice. The reason is a dramatic discrepancy between the practical needs of project appraisal and features of formal EcoRA methodology.

The formal EcoRA focuses on relatively manageable and observable biological units (individual animals or plants or small populations of these organisms) rather than on the ecosystems. In turn, EIA is mostly concerned with ecosystem protection and with cases of endangered species that can potentially be affected.

In this framework hazard assessment is mainly based on toxicity testing in clean laboratory conditions. Findings of laboratory studies are then extrapolated to higher levels of natural system hierarchy (from organisms to communities and even ecosystems) using various factors (Smrchek and Zeeman, 1998).

For this "bottom–up" approach to ecosystem assessment a methodological framework has been rapidly developed: for a number of chemical and test organisms, substantive databases on species toxicity are already established, safety and uncertainty factors has been determined; testing schemes, exposure models, and algorithms for risk estimation and evaluation has been elaborated.

However, applicability of the bottom–up approach is limited primarily by cost implications: to conduct ecosystem risk assessment following accurately the formal U.S. EPA procedure, an assessor must spent huge amounts of time and money on collecting necessary input data, data processing and interpreting the outputs. Of importance, very specific data are often required that cannot be easily obtained with the help of standard environmental monitoring studies.

The "top–down" approach to ecosystem assessment that considers an elementary ecosystem as a receptor for evaluating toxic effects is currently in the making and remains difficult to carry out. One can identify the following key problem spots in methodology for ecosystem risk assessment:

- selecting appropriate assessment endpoints (at present, a number of them are already proposed including ecosystem integrity, biodiversity, resilience, sustainability (Lohani et al., 1997); however, many of these concepts are hardly applied to practical needs);
- deriving numerical criteria assessing state and effects on ecosystems (measurement endpoints);
- developing predictive tools (firstly biogeochemical models) for describing ecosystems behavior and their validation;
- establishing the assessment benchmarks ("unpolluted" ecosystems of particular type);
- establishing and justifying risk mitigation strategies (defining a threshold values for stressor impacts).

Due to lack of established, user-friendly, and cost effective quantitative approaches to ecosystem risk assessment in EcoRA, in the current EIA practice of project appraisal ecosystem risk assessment (if conducted) is usually comparative or qualitative (see, e.g., Lohani et al. (1997) for in-depth discussion).

Qualitative findings of ecosystem risk assessments are of low utility for risk management. They cannot be compared with quantitative estimates of other risks; this compromises the ability of risk ranking to provide insights to setting priorities. It is particularly difficult to convert them into a format applicable for cost–benefit analysis, which is a key tool that any proponent uses in decision-making on a proposed project.

The authors believe there is an obvious need for improving methodology for assessing ecosystem risks. It seems reasonable to review existing approaches to quantitative assessment of ecological effects, which are not usually included in the EcoRA domain. One promising solution may be the Critical Load and Level (CLL) methodology. Its key features and potential applicability to ecosystem risk assessment are discussed in the following section.

5. CRITICAL LOAD AND LEVEL (CLL) APPROACH FOR ASSESSMENT OF ECOSYSTEM RISKS

As has been mentioned above, the CLL concept was introduced initially for emission control at an international scale under the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP). From the beginning it has been applied for regional and local assessments of ecological effects (see, e.g., Bashkin et al., 2002; Ignatova, 2003; Priputina and Mikhailov, 2003). The latest advances and trends in developing the CLL concept encouraged researchers to consider if critical loads and their exceedances could be applied in EIA for assessing effects on ecosystems.

Critical loads and levels are measurable quantitative estimates showing the degree of tolerable exposure of receptors to one or more pollutants. According to present knowledge, when this exposure remains below the critical load and level thresholds, significant harmful effects on specified receptors do not occur (Gregor, 2003). They serve as reference points against which pollution levels can be compared and potential risks to environmental components can be estimated.

The most common shortcomings include:

- failure to analyze impacts beyond development site boundaries,
- failure to quantify ecological impacts (vague descriptive predictions are the norm),
- failure to identify or measure cumulative ecological effects,
- failure to mitigate important ecological impacts (proposed mitigation measures are inappropriate and implementation is not mandatory),
- lack of monitoring or follow-up (actual outcomes are not known and no corrective action can be taken, e.g., in the event of mitigation failure).

The critical Load and Level (CLL) concept is an important element for emission control policies in Europe. It has become the internationally agreed scientific underpinning for setting targets in controlling SO_2, NO_x, and NH_3 emissions; development of critical loads and levels and similar pollution abatement strategies for heavy metals and persistent organic pollutants (POPs) is currently in the making (Bashkin, 2002).

Initially, the United Nations Economic Commission for Europe (UNECE) introduced the CLL approach into the control of transboundary air pollution under the Convention on Long-Range Transboundary Air Pollution (CLRTAP). In 1994, critical loads of acidity served as inputs to the second Sulphur Protocol. More recently, European critical load maps were central to the development of the Gothenburg Protocol on acidification, eutrophication and ground level ozone adopted by the Executive Body of the UNECE CLRTAP in November 1999. Critical load calculating and mapping has been currently undertaken worldwide at national levels including countries, which are not bound with CLRTAP obligations, e.g., India, China, Thailand (Bashkin and Park, 1998; Bashkin, 2002, 2003; Satsangi et al., 2003; Ye et al., 2002).

Over time, there has been growing interest in defining critical loads at a regional level to define sensitivity of particular areas to inputs of pollutants and to set specific threshold exposure values (see, e.g., Lien et al., 1995; Henriksen et al., 2002; Craenen et al., 2000; Helliwell and Kernan, 2004). Most of the research on critical loads and levels is concentrated in regions sensitive to sulfur and nitrogen pollution to generate input data for mapping critical loads and levels following common methodology developed under the Convention framework.

More and more research publications on critical loads of acidity for specific lakes, their catchments, or forest ecosystems within defined boundaries are appearing. However, local studies to provide comprehensive input data on biogeochemical parameters for CLL estimations are not being undertaken in countries where environmental monitoring network is rare. For example, Bashkin et al. (2002) proposed an approach to defining critical loads and their exceedances of nitrogen, sulfur and heavy metals for ecosystems adjacent to the Yamal-Zapad gas pipeline located in the North of European Russia (see Bashkin et al. (2002), and Chapter 20 of this book). Ignatova (2003) discussed findings of calculating and mapping critical loads and levels of acidifying pollutants for a small catchment and three monitoring sites in Bulgaria. Priputina and Mikhailov (2003) applied the CCL approach to calculating critical loads for heavy metals for forest ecosystems in European Russia.

The latest advances and trends in developing the CLL concept, which has been constantly validated and progressively improved since its international authorization in 1988 (Cresser, 2000), have encouraged the author to consider if critical loads and their exceedances could be used in EIA for assessing effects on ecosystems.

The following strengths of the CLL approach in the context of EcoRA/EIA are summarized below.

Quantitative nature of the CLL approach. Numerical tolerable exposure levels for pollutants of concern are defined to establish quantitative thresholds for risk characterization; therefore the CLL approach provides a basis for quantitative ecosystem risk and damage assessment.

Scope of the CLL approach. Critical loads and levels can be calculated for various specified "sensitive elements of the environment" (UNECE CLRTAP 2004, V-1). However, terrestrial and aquatic ecosystems are most frequently referred to as receptors in this effect-based approach. In addition, specific parts of ecosystems (e.g., populations of most valuable species) or ecosystem characteristics can be defined as receptors as well (UNECE CLRTAP, 2004). Such flexibility and established provisions for ecosystem assessment makes the CLL concept a promising solution for ecosystem risk assessment and a potential substitute for site-specific chemical RA following the bottom–up approach.

CLL approach and ecosystem, risk analysis. This approach provides insights on assessment and measurement endpoints for ecosystem-level EcoRA since it has a set of environmental criteria to detect the state of ecosystems; critical load itself can be treated as a criterion for ecosystem sustainability (Bashkin, 2002). Moreover, one can derive "spatial" ecosystem risk estimates based on the percentage of ecosystems protected/potentially at risk under the current and predicted level of pollutant loads.

CLL approach and EIA baselines studies. While calculating and mapping critical loads, an assessor reviews and systematize most of the data on current state of environment in the site vicinity; the clear and illustrative picture of receptors and their sensitivity to potential impacts is an output of this process.

CLL approach and impact mitigation. Critical loads are particularly useful for elaborating more focused and impact-oriented environmental monitoring programs; mapping critical loads and their exceedances highlights ecosystems (or areas) being damaged by actual or potential pollutant loads giving hints on siting environmental monitoring locations. In turn, critical levels provide a basis for defining maximum permissible emissions to substantiate the development of mitigation measures.

CLL mapping is extremely useful in *communicating* findings of environmental impact studies both for general public and decision-makers.

Input data requirements. Critical loads and levels are estimated with help of biogeochemical models that require a great deal of input data on parameters of biogeochemical turnover and pollutant cycling in ecosystems. Ideally, an assessor should use findings of field studies aimed at measuring all necessary parameters with appropriate extent of accuracy and at appropriate scale. For regions with underdeveloped networks of environmental monitoring (like vast areas of the Russian Federation or China), lack of required data would be a key obstacle for applying CLL within EIA. At the same time, simplified algorithms for CLL calculation have already been elaborated. One of these methods allows for defining critical loads through internal ecosystem characteristics and derived environmental criteria including soil properties, vegetation type, climatic data (Bashkin et al., 1995, see also Chapter 2). Therefore, an assessor is able to select a CL algorithm among those available bearing in mind input data availability (both empirical, modeled, and literature data) and selected highest degree of uncertainty.

Credibility of the CLL approach is relatively high: today the CLL approach is a widely-known internationally agreed effect-oriented methodology applied worldwide; this aspect is meaningful in communicating research findings on effects and making decisions on risk management.

Progressive update and improvement. Even those who criticize the theoretical soundness of this approach (Skeffington, 1999; Cresser, 2000) acknowledge efforts to validate and improve the CLL concept for increasing degree of confidence of critical loads and levels. UNECE CLRTAP provided an organizational and scientific framework for CLL elaboration having established a program dealing with collecting input data for the CLL calculation (EMEP), and a number of programs under the Working Group of Effects (WGE) focused on processing collected data while calculating CLL for specific receptors (forest ecosystems, aquatic ecosystems, human health, materials) as well as respective International Cooperative Programs (ICPs). In addition, there are ICPs engaged in developing methodologies and improving practice of mapping and modeling and environmental monitoring (Gregor, 2003). The recent trend in developing CLL methodology is introducing a dynamic approach into modeling (see Chapter 6 of Modelling and Mapping Manual (2004) for details).

Usability of CLL. There are plenty of practical guidelines on calculating critical loads and levels including the constantly updated Manual on Methodologies and Criteria for Modeling and Mapping Critical Loads & Levels and Air Pollution Effects, Risks and Trends (Modelling and Mapping Manual, 2004)[3]. Moreover, many research groups engaged in biogeochemical model development make them available as "freeware". Annual reports published by the National Focal Centers of the LRTAP Convention as provides insights on methodologies and partially input data for the CLL calculations.

The key shortcoming of the CLL approach from an EIA practitioner's perspective is data uncertainty—a "sore subject" for any predictive exercise. This is especially true for a simplified algorithm for critical load calculation (see above). Both assessors and reviewers will ask the following questions:

- Do critical loads really protect ecosystem health?
- Do applied models provide scientifically defensive results?
- Are current models capable of acceptive relevant data?

The uncertainty analysis that is a part of formal EcoRA methodology is designed to ensure adequate estimation of ecological effects based on a state-of-the-art scientific basis. Moreover, if applied on a local scale for site-specific assessments, with the use of empirical input data as biogeochemical parameters, the CLL approach is likely to provide results with a higher degree of confidence than the formal EcoRA model.

In the authors' opinion, even if imperfect, the CLL approach is preferable to apply for ecosystem risk assessment than a qualitative EcoRA based mainly on expert judgment.

In response to the need for more consistent treatment of ecological effects resulting from development projects, the current paper proposes a structured framework for introducing the CLL concept as an approach to ecosystem risk assessment into EIA. The model of the "integrated" process depicted in Figure 3 represents the widely accepted idea of "embedding" risk assessment into EIA (Arquiaga et al., 1992; NATO/CCMS 1997; Poborski, 1999; Demidova and Cherp, 2004). It is organized according to the sequence of generic EIA stages: *screening, scoping, impact prediction and evaluation, mitigation, reporting, decision-making*, and *post-project monitoring and evaluation (EIA follow-up)* with *public participation* and *consideration of alternatives* potentially incorporated at all stages of the process (Wood, 1995; Canter, 1996; Lee and George, 2000). The CLL methodology is considered as a quantitative approach to assessing ecological effects. Proposed CLL inputs into the EIA process are discussed below.

In the proposed model project, appraisal starts with addressing two questions at the screening stage:

- Is EIA necessary? and
- Is EcoRA within EIA necessary?

[3] The Mapping Manual is available via Internet at www.icpmodelling.org.

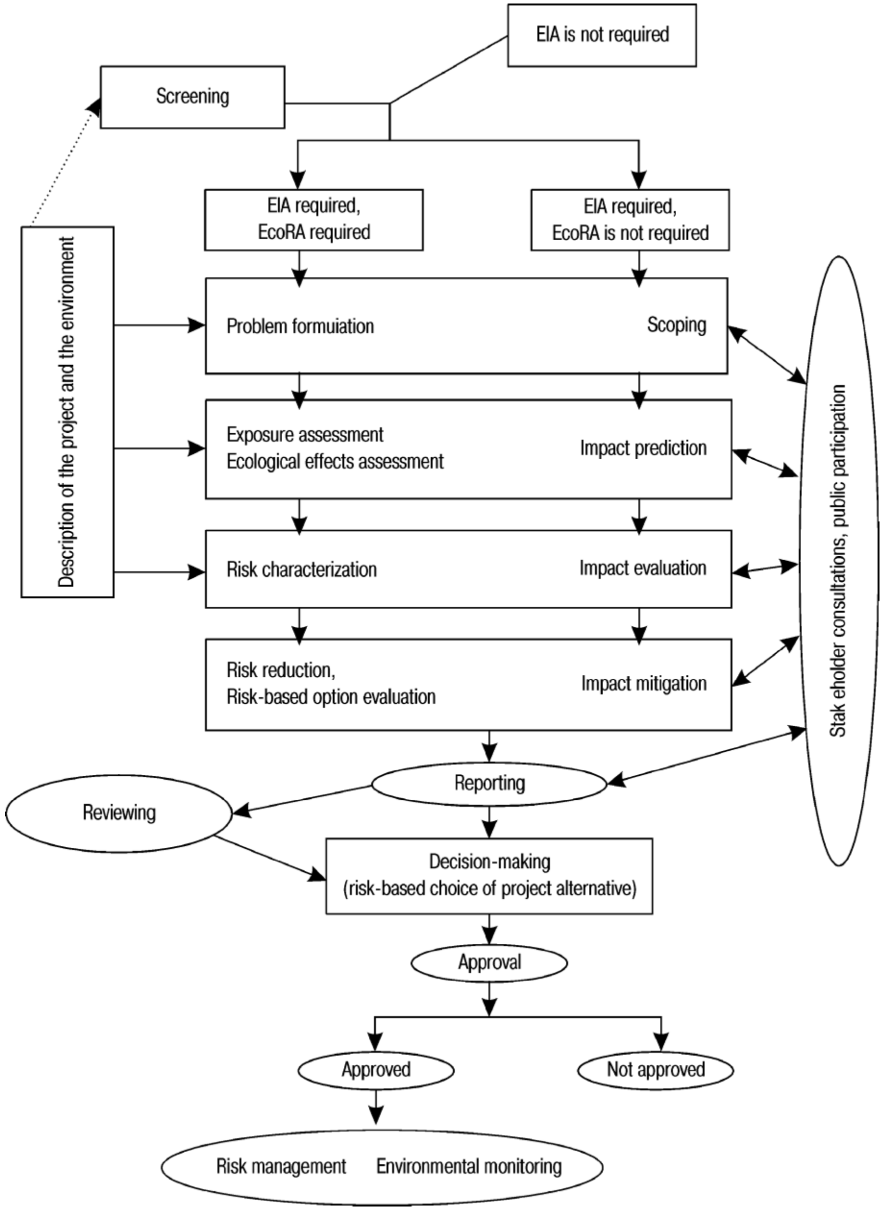

Figure 3. The model for assessment of ecosystem risks in the EIA for projects with significant ecological implications.

It is the responsibility of the appointed environmental consultants to undertake preliminary investigations and decide if a proposed development may result in significant ecological effects. Data on "risk agents" including ecosystem stressors associated with the project and their potential impacts on the environment underpin screening decisions.

Scoping should include defining project alternatives, compiling the list of project impacts, which should be subject to comprehensive impact assessment and planning the further steps of the assessment process. In the formal EcoRA framework this step is related to problem formulation. A separate task of this stage is to select methods and procedures for dealing with particular impacts. For ecosystem effects available information on stressors, effects, and receptors is analyzed to define risk assessment endpoints (assessment and measurement endpoints) and possible conceptual models. In addition, policy and regulatory requirements, available budget and an acceptable level of uncertainty are considered in developing a plan for EcoRA. Here the assessment team may consider applicability of the CLL concept to project ecological effects and develop a plan of specific studies for calculating and mapping critical loads. The outcome of the scoping is to be an EIA Terms of Reference (ToR) referring to all abovementioned issues.

The next step is impact prediction that requires detailed quantitative information about the sources of risk agents, exposure models, the receptors and possible changes in the state of these receptors caused by the defined agents. If the CLL concept was selected for assessment ecosystem effects it should firstly be utilized for impact baseline studies or assessing the "do-nothing" scenario. In this context CLL calculation includes the following steps (Bashkin, 2002):

- characterizing receptors that are potentially affected by the proposed development,
- defining environmental quality criteria,
- collecting input data for CLL calculations,
- calculating critical loads (CLs),
- comparing CLs with actual loads to calculate the exceedances.

When the environmental baseline is established one can proceed with predicting the magnitude of potential impacts onto receptors at risk for *exposure assessment* in EcoRA terms. This includes:

- quantifying emissions of pollutants of concern,
- modeling their transport in the environmental media,
- estimating the predicted exposure levels,
- estimating predicted loads.

Under the CLL approach, *ecosystem effect assessment* means comparing critical loads with predicted loads of pollutants. Of importance, this may be limited to an ecosystem as a whole without further evaluating adverse effects on specific ecosystem components. CL mapping with help of GIS is especially useful for this purpose.

Impact prediction should cover all project alternatives selected at scoping (either spatial or technological) and project phases (construction, operation, closure and post-closure are the main subdivisions). Moreover, exposure assessment should cover both normal operation and accidental conditions.

Significance of the predicted impacts should be assessed in the process of impact evaluation or interpretation. At this stage the health risk estimates (quantitative and qualitative) are analyzed in terms of their acceptability against relevant regulatory and/or technical criteria: environmental quality standards or exposure limits.

Critical load exceedances may serve as the basis for interpreting ecological impacts as ecological risks (or rather changes in the level of current risk to "ecosystem health"). This would refer to the process of ecological risk characterization.

There are a number of approaches to measuring risks depending on assessment and measurement endpoints selected. At ecosystem level, one can propose a percentage of the affected area with CLs exceeded as an acceptable quantitative parameter for ecosystem risk magnitude. In pristine areas, actual state of the environment may be taken as a reference point for risk characterization.

As to risk significance, the degree of alteration in the current environment should be amended with qualitative and semi-qualitative criteria. Ecological impact significance should be considered in terms of:

- ecosystem resilience to particular impacts,
- principal reversibility of potential ecosystem damage,
- threats to valuable ecosystem components, etc.

The estimation of accuracy of quantitative predictions and the degree of uncertainty of the assessment findings should be attempted as well.

The results of impact prediction and evaluation are used for designing impact mitigation measures that aim to prevent or reduce the adverse effects associated with the projects and restore or compensate the predicted damage to the environment. Impact mitigation should firstly involve risk reduction measures: (1) control of the source of risk agents; (2) control of the exposure; (3) administrative/managerial improvements; (4) risk communication allowing for more comprehensive risk perception. The selection of appropriate mitigation measured would benefit from using risk–benefit analysis (with formal quantification of residual risks for every option if applicable).

Following the logic of the CLL approach, impact mitigation in EIA is to derive critical limits of exposure (concentrations of pollutants in exposure media) and based on these values calculating maximum permissible emissions that ecosystems in the site vicinity would sustain during the life-time of the proposed facility. Therefore, any technology that allows for not exceeding CLs for potentially affected ecosystems should be acceptable from the environmental viewpoint, not exclusively the Best Available Technology (BAT) as often recommended.

6. UNCERTAINTY IN IRA AND ERA CALCULATIONS

One can identify two major categories of uncertainty in EIA: data (scientific) uncertainty inherited in input data (e.g., incomplete or irrelevant baseline information, project characteristics, the misidentification of sources of impacts, as well as secondary, and cumulative impacts) and in impact prediction based on these data (lack of scientific evidence on the nature of affected objects and impacts, the misidentification of source–pathway–receptor relationships, model errors, misuse of proxy data from the analogous contexts); and decision (societal) uncertainty resulting from, e.g., inadequate scoping of impacts, imperfection of impact evaluation (e.g., insufficient provisions for public participation), "human factor" in formal decision-making (e.g., subjectivity, bias, any kind of pressure on a decision-maker), lack of strategic plans and policies and possible implications of nearby developments (Demidova, 2002).

Some consequences of increased pollution of air, water and soil occur abruptly or over a short period of time. Such is the case, for instance, with the outbreak of pollution-induced diseases, or the collapse of an ecosystem as one of its links ceases to perform. Avoiding or preparing for such catastrophes is particularly difficult when occurrence conditions involve uncertainty.

In spite of almost global attraction of the critical load concept, the quantitative assessment of critical load values is connected till now with some uncertainties. The phrase "significant harmful effects" in the definition of critical load is of course susceptible to interpretation, depending on the kind of effects considered and the amount of harm accepted (De Vries and Bakker, 1998a, 1998b). Regarding the effects considered in terrestrial ecosystems, a distinction can be made in effects on:

- soil microorganisms and soil fauna responsible for biogeochemical cycling in soil (e.g., decreased biodiversity);
- vascular plants including crops in agricultural soils and trees in forest soils (e.g., bioproductivity losses);
- terrestrial fauna such as animals and birds (e.g., reproduction decrease);
- human beings as a final consumer in biogeochemical food webs (e.g., increasing migration of heavy metals due to soil acidification with exceeding acceptable human daily intake, etc.).

In aquatic ecosystems, it is necessary to consider the whole biogeochemical structure of these communities and a distinction can be made accounting for the diversity of food webs:

- aquatic and benthic organisms (decreased productivity and biodiversity);
- aquatic plants (e.g., decreased biodiversity, eutrophication);

- human beings who consume fish or drinking water (surface water) contaminated with mobile forms of heavy metals due to acidification processes (e.g., poisoning and death).

7. BENEFITS OF APPLYING CCL IN EIA

Therefore, the CLL concept is a valuable methodology for ecological impact and risk assessment and is easily adjusted to the formal EIA procedure. The proposed framework could be applied to EIAs of development projects with high ecological implications that can potentially affect the environment both on local and regional scales. The model may be applicable to developments that involve releases of acidifying and eutrofying compounds, heavy metals and POPs into the environment in areas with high ecosystem vulnerability and/or pristine areas.

Ecological effects are often treated inadequately in the assessment of environmental impacts of proposed developments, while lack of quantitative ecological impact predictions is mentioned among key drawbacks of the current EIA practice. The idea of integrating RA into EIA for improving the quality of EI studies has been supported by many EIA practitioners. At the same time, formal ecological risk assessment has significant limitations for assessing ecosystems risks related to proposed developments.

To improve addressing ecological implications of human activities, the author has attempted to incorporate the Critical Load and Level (CLL) approach, an established methodology for assessing effects of industrial pollution on ecosystems and their sensitive components, into the EIA process. Benefits of and obstacles to applying that approach to assessing ecosystem effects within EIA were analyzed. Finally, a structured framework for CLL application for ecosystem risk assessment in EIA aimed at integrating three assessment tools was presented and key CLL inputs into impact assessment stages were discussed.

The proposed model of integrated assessment process is suggested for testing in EIAs for development projects with high ecological implications: those associated with releases of pollutants covered by current CLL calculating and mapping methodology and located in areas particularly sensitive to the selected indicator chemicals.