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Role of Modelling in Ecological Risk Assessment and Management with Emphasis on the Offshore Oil and Gas Industry

Rôle de la modélisation dans l'évaluation et la gestion des dangers pour l'écologie, en particulier pour le secteur des hydrocarbures extracôtiers

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ABSTRACT

Mathematical models have become indispensable tools in Ecological Risk Assessment (ERA) and Ecological Risk Management (ERM). The human mind cannot deal with the multifold of interactions that can be encompassed by models. Modern environmental management must be based on the quantitative forecasts of models. That said, models do not capture reality. At best, they only capture the essence of reality, an essence that must be specified in the model. Furthermore, the gap between reality and models ensures that a variety of competing models may be available. Misunderstanding, misapplication, incorrect structural forms of modelled processes, and erroneous parameter estimates can lead to misrepresentation of reality in models and consequent mistakes in assessment and management. Hence, models must be tested against reality and used by those that have expertise in the reality being modelled. In addition, a range of available models and parameter values should be evaluated to probe the gap between models and reality.

ERA has been defined (Suter, 1993) as "the process of assigning magnitudes and probabilities to the adverse effects of human activities or natural catastrophes." ERM has been defined (Pittinger *et al*., 1998) as "the process of identifying, evaluating, selecting, and implementing cost-effective, integrated actions that manage risks to environmental systems while emphasising scientific, social, economic, cultural, technological feasibility, political, and legal considerations." ERA is the realm of the science-grounded assessor who seeks to provide objective assessments of risk, whereas ERM is the realm of the informed manager who seeks to provide balanced decisions in reducing risk. The risk assessor must know why the assessment is being done and the decisions that have to be made in order to provide answers useful to the risk manager. On the other hand the risk assessor must be free of management pressure to give predestined results.

Several paradigm shifts are looming that may greatly complicate ERA and ERM over the next decade, which may be the period leading up to the establishment of an oil and gas industry of the B.C. coast. A shift is occurring in chemical toxicology from estimating the probability of effects based on exposure medium to that based on body residue. This shift is driven to some extent by the inherent toxicity concept that holds that noncarcinogenic modes of toxicity seem to be associated with different body residue ranges of chemical toxicants within organisms. This shift also divorces uptake processes from toxic effect by moving a step closer to the site of action, i.e., from the medium (e.g., water, sediment) to the body. Another paradigm shift, also in toxicology, is driven by the re-ascendancy from the late 19^{th} and early 20^{th} centuries of hormesis, which is challenging the orthodoxy of the linear, non-threshold dose-response model in ERA. Hormesis, which must be distinguished from homeopathy, is a well-documented phenomenon in which low doses of a toxicant confer a benefit to an organism. Also returning from relative obscurity for most of the 20^{th} century is Bayesian statistics. Bayesian statistics estimates the probability that a hypothesis is true, given the evidence. In the case of ERM, Bayesian statistics provides useful information about the degree of belief in different hypotheses that allows decision-makers to choose among competing outcomes. Frequentist statistics, which have held sway for the about seven decades, cannot provide such easy to understand output. Instead, these statistics are limited to probability of obtaining the evidence, given that the null hypothesis is true.

Decision analysis is a process for helping make complex decisions. The basic tool of the decision analysis process is the decision tree, which allows a complex problem to be decomposed into its component parts and recomposed in a coherent and consistent manner. Decision analysis like ERA is a service to the decision-maker

RÉSUMÉ

Les modèles mathématiques sont devenus des outils indispensables pour l'évaluation et la gestion des dangers pour l'écologie. Comme notre cerveau n'a pas la capacité de traiter les multiples interactions qui peuvent être incluses dans les modèles, la gestion moderne de l'environnement doit reposer sur les prévisions quantitatives tirées de ceux-ci. Cela dit, les modèles ne rendent pas la réalité. Au mieux, ils ne reproduisent que l'essence de la réalité, essence qui doit y être précisée. De plus, l'écart entre la réalité et les modèles signifie qu'une gamme de modèles concurrents peuvent être utilisés. Une mauvaise compréhension, un mauvais emploi, des formes structurelles inexactes des processus modélisés et des estimations erronées des paramètres peuvent mener à la représentation erronée de la réalité dans les modèles et à des erreurs conséquentes dans l'évaluation et la gestion. Les modèles doivent donc être confrontés à la réalité et utilisés

par ceux qui connaissent bien la réalité modélisée. En outre, une gamme de modèles et de valeurs de paramètres disponibles devraient être évalués afin d'établir l'écart entre les modèles et la réalité.

L'évaluation des dangers pour l'écologie (EDE) a été définie comme le processus d'attribution de grandeurs et de probabilités aux effets nuisibles des activités humaines ou des catastrophes naturelles (Suter, 1993), et la gestion des dangers pour l'écologie (GDE) comme le processus d'identification, d'évaluation, de choix et de mise en oeuvre de mesures intégrées et efficaces en terme de coûts permettant de gérer les risques pour les systèmes environnementaux tout en mettant accent sur des considérations de faisabilité technologique et d'ordre scientifique, social, économique, culturel, politique et juridique (Pittinger *et al*., 1998). L'EDE est le domaine de l'évaluateur versé dans les sciences, qui tente de fournir des évaluations objectives des dangers, tandis que la GDE est le domaine du gestionnaire informé qui tente de prendre des décisions éclairées pour réduire les dangers. L'évaluateur des risques doit savoir pourquoi l'évaluation est faite et quelles décisions doivent être prises pour être en mesure de fournir des réponses utiles au gestionnaire des risques. Par contre, il ne doit pas avoir à composer avec des pressions de la part des gestionnaires pour fournir des résultats prédéterminés.

Les changements de paradigme qui paraissent imminents pourraient fortement compliquer l'EDE et la GDE au cours de la prochaine décennie, qui pourrait être la période qui verra l'établissement d'un secteur des hydrocarbures au large des côtes de la Colombie-Britannique. La toxicologie des produits chimiques est en voie de prendre un virage, de l'estimation de la probabilité des effets reposant sur le milieu d'exposition à l'estimation reposant sur les résidus dans les tissus. Ce virage est piloté dans une certaine mesure par le concept inhérent à la toxicité voulant que les modes non carcinogènes de toxicité semblent liés à différentes gammes de résidus de substances chimiques toxiques dans les organismes. Il sépare en outre les processus d'absorption de l'effet toxique en se rapprochant du site d'action, c.-à-d., du milieu (p. ex., eau, sédiments) au corps. Un autre changement de paradigme, en toxicologie à nouveau, est piloté par l'importance renouvelée donnée à l'hormèse, découverte à la fin du XIX^{e} et au début du XX^{e} siècles, qui met au défi l'orthodoxie du modèle linéaire de la dose sans seuil d'exposition et de la réaction de l'EDE. L'hormèse, qu'il ne faut pas confondre avec l'homéopathie, est un phénomène bien documenté qui veut que certaines substances toxiques administrées à faibles doses ont une action stimulante sur un organisme. Les statistiques bayésiennes sortent aussi des ténèbres où elles étaient enfouies pendant la plus grande partie du XX^{e} siècle. Ces statistiques permettent d'estimer la probabilité qu'une hypothèse est vraie à la lumière des faits. Dans le cas de la GDE, elles permettent d'obtenir de l'information utile sur le niveau de croyance dans les diverses hypothèses, ce qui permet aux décideurs de choisir entre les résultats opposés. Les statistiques des fréquences, qui ont régné pendant environ sept décennies, sont incapables de donner des résultats aussi faciles à comprendre, car elles se limitent à établir la probabilité d'obtenir la preuve, étant donné que l'hypothèse nulle est vraie.

L'analyse de décision est un processus permettant de prendre des décisions complexes, dont l'outil de base est l'arbre de décision. Celui-ci permet de décomposer un problème complexe et de le reconstituer d'une manière cohérente et constante. L'analyse de décision, tout comme l'EDE, est un service à la disposition des décideurs.

INTRODUCTION

The process of risk assessment and its application to fisheries management has evolved since the early 1990s. There has been much variation in terminology, and Francis and Shotton (1997) recently published a clarifying review of the development of risk assessment in fisheries. They suggested that there is consensus about a two-stage decision making process that includes: 1) formulation of advice for fisheries managers in a way that conveys the possible consequences of uncertainty and provides an evaluation of the expected scientific effects of alternative management options ("risk assessment") and 2) a way of combining probabilities of different states of the system with the magnitude of the indicators and with sociological, monetary and other factors in making decisions ("risk management"). The first stage has become relatively formal in execution, differing only in technical details, but the second stage is usually informal, subjective and often unrelated to the first stage (Francis and Shotton, 1997).

Risk is often erroneously confused with hazard. Risk can be defined as the probability that a specified effect will occur from a hazard (Suter, 1993). A more expansive definition of risk is the probability of threat (hazard) times the probability of exposure with an expected, quantifiable effect, or for short: risk = threat x exposure-effect. Kaplan and Garrick (1981{#949}) provide a formal mathematical definition of risk. With risk comes the concept of uncertainty and how it influences the decision-making process. Risk management is the process that determines what actions will be taken in response to a specified risk. There are several tools in the risk management toolbox (Hilborn *et al*., 2001), but an ERA is one of the most important tools a risk manager uses to make critical decisions. ERAs present, quantify, and prioritise the level of risk, improve the understanding of risk, and, importantly, highlight the levels of error and gaps in knowledge. The purpose of the ERA is to assist in the protection and management of the environment through use of objective evaluation procedures that use credible scientific information (Suter, 1993).

Ecological assessments (EAs) and ERAs have evolved over the last two decades and grew out of the process of environmental impact assessments. ERAs also have historical roots in the fields of human risk assessments. ERM has evolved from risk management associated with societal concerns, including economic, social, and ecological.

The fields of EA, ERA and ERM are complicated by jargon and, at times, inconsistent definitions. A full discussion of these fields is beyond the scope of this paper and can be found in other sources such as the recent US EPA guidelines document (US EPA, 1998) and Suter's book (1993). The purpose of this paper is to present key concepts and definitions, to point out potential pitfalls in the EA and ERA processes, and to show how these concepts tie into the field of risk management and the decision-making process. In addition, this paper provides a short review of modelling tools and assessment techniques that pertain to oil and gas exploration, production, and spills. Finally, this paper discusses some emerging issues that must be considered in future ERAs and ERM decisions.

ECOLOGICAL ASSESSMENT AND ECOLOGICAL RISK ASSESSMENT

THE DIFFERENCE BETWEEN ECOLOGICAL ASSESSMENT AND ECOLOGICAL RISK ASSESSMENT

EA and ERA are sometimes used synonymously, but are, in-fact, different processes with different goals (for discussions see Pittinger *et al*, 2000; Douben, 2000; Benson *et al*., 2000). An EA describes and characterises the biological, physical and chemical parameters of a region or ecosystem. It may identify potential hazards associated with some stressor to the environment, but, generally, it does not attempt to specify and quantify those risks. EAs may involve the development and use of indicators to look at trends and tend to be investigatory without dealing with cause/effect relationships (Pittinger *et al*., 1998).

ERAs evaluate the likelihood that a range of ecological effects are occurring or may occur, some of which are adverse and some of which are not, because of exposure to one or more stressors (US EPA, 1998). Suter (1993) defines the ERA as “the process of assigning magnitudes and probabilities to the adverse effects of human activities or natural catastrophes.” The process seeks to build causal relationships between stressor exposure and receptor response (Pittinger *et al*. 2000). Often an EA is a component of or stimulus for a full ERA, but they are separate entities.

The ERA process was historically developed to deal with ecosystem responses to chemical/contaminant stressors. Practitioners recently have looked beyond chemical stressors to biological (e.g. exotic species) or physical (e.g. climate change) stressors. The appropriate tools to deal with these additional types of stressors have not been well developed (Pittinger *et al*., 2000; Benson *et al*., 2000).

Variability and uncertainty are inherent in the ERA process. Variability is defined as “the distribution (heterogeneity, diversity, inter-individual differences, etc.) of values of a quantified variable that characterises members of a well specified target population" (Kelly *et al*., 1997). Kelly *et al*. (1997) prefer the term "target population variability" instead of just "variability," in which statisticians often include sampling and measurement errors that are part of uncertainty in the ERA context. Uncertainty is the estimated distribution from all the uncertain knowledge. There are generally three sources of uncertainty: random variation, incomplete or incorrect scientific knowledge, and human error. For example, sources of uncertainty may include an incomplete understanding a chemical's environmental fate and lack of knowledge about the dose-response relationship for a species in question.

All data have uncertainty and limitations. Laboratory studies may be highly controlled, but not represent real-world scenarios. Field studies may be highly variable and lack statistical rigor. One critical aspect of uncertainty appears to be the meaning of dose within the contexts of constant exposure and pulse exposure (for discussions see Glasby, 2000; Lee and Jones-Lee, 1999; Scholz and Collier, 2000). Other aspects of uncertainty with regard to data include the use of single stressor data when assessing multiple stressor systems.

With respect to human error, inappropriate data choices should be considered. For example, forecasts based on dose-response and fate data for a chemical studied under temperate conditions may vary widely from reality under cold conditions (for discussion see Robins and Sevigny, 2000). Chemical fate and bioavailability may differ among freshwater, estuarine, and fully marine conditions.

It is vital to a credible ERA that the assessor knows and communicates the level of uncertainty associated with each step in the process and the sources of scientific information used. The US EPA framework (1998) outlines uncertainty issues with ERAs.

THE ERA PROCESS

The ERA process is generally a step-by-step process. In addition to various phases, the process is also often divided into a hierarchy of tiers[1], each assessing different levels of uncertainty in an attempt to accurately quantify risk. The first tier is often referred to as the screening-level risk assessment (SLRA). It tends to be broad-based to ensure that all the important issues are identified and considered. It is often extremely conservative in its levels of uncertainty[2] and variability to ensure the risks are not understated (Hill *et al*., 2000). Upper-level tiers are frequently referred to as detailed-level risk assessments (DLRA) and attempt to provide real-world scenarios by reducing residual conservatism arising from use of 10-fold or other inflated uncertainty factors (Hill *et al*., 2000). Although it is generally agreed that ERAs should target large organisational structures such as populations (Suter, 1993; Durda and Preziosi, 1999), Hill *et al*. (2000) suggest that SLRAs, because of their high level of conservatism, may target the stressor/response relationship between individual members of a populations and individual stressors. Although this approach may overestimate risk (Bonnell and Atkinson, 1999), the tools to deal with populations at the SLRA may not be available. Hill *et al*. (2000) suggest that a DLRA is more appropriate to deal with single and multiple stressors and responses to them in populations.

It is generally agreed that in the lower level tiers (e.g. SLRA) there are three (sometimes four) phases: problem formulation, analysis, and risk characterisation. These phases are reviewed in a US EPA guidelines document (US EPA, 1998), but briefly, problem formulation involves determining goals, selecting assessment endpoints, preparing models and developing the analytical plan. The analysis phase involves the evaluation of stressor exposure on the environment/receptor and the determination of cause and effect relationships (e.g. species dose-response toxicity studies). This phase is sometimes also referred to as exposure-response assessment (Barnthouse, 1992). The risk characterisation phase develops an estimate of the risk type and level by integrating the exposure profiles. Communicating this information facilitates the decision-making process. Adaptations to this three-phase system do occur, but the process remains highly similar.

A critical step involved in the problem formulation phase is the assignment of relevant assessment endpoints. The US EPA (1998) asserts that "[a]ssessment endpoints are explicit expressions of the actual environmental value that is to be protected, operationally defined by an ecological identity and its attributes." Selecting assessment endpoints identifies what is at risk. Suter (1993) points out that endpoints should be

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1 A cumulative, three-tiered approach to ERA is favoured by the US Navy (http://newweb.ead.anl.gov/ecorisk/process/pdf/index.cfm), based on US EPA guidelines for site assessment under the Superfund Program. The tiers are: screening risk assessment (Tier 1), baseline ERA, which includes detailed assessment of exposure and hazard (Tier 2), and evaluation of remedial alternatives that may include quantitative evaluations (Tier 3). Tier 1 includes a site visit, exposure estimates and a risk calculation. For the development of marine water quality criteria the US EPA uses cumulative, tiered biological surveys that "range from 0 to 4 in complexity with Tier 0 being a simple desktop screening assessment and Tier 4 being the most rigorous and involving four or more field site visits" (Russo, 2002).

2 At the screening level 10-fold uncertainty factors are often used to account for unknowns in extrapolating from single-species laboratory-toxicity data to ecosystems, although Bonnell and Atkinson (1999) have proposed an alternative approach that is based on quantifying total certainty before calculating uncertainty. Their approach is still very conservative to exclude false negatives (i.e., not rejecting the hypothesis that a chemical will have no effect (null hypothesis), when that hypothesis is false) and merits from being more comprehensive, objective, transparent and traceable than the 10-fold uncertainty factor tradition.

relevant to both the biological system and society, must be workable (for measurements purposes) and should be sensitive to the stressor in question.

One area of confusion seems to be the difference between endpoints and management goals. The US EPA guidelines (1998) state that it is important that assessment endpoints are not confused with management goals, as endpoints should not imply any sense of protection, maintenance, or increase/decrease in some target. The assessment endpoint is an entity, and it has as an important attribute the need to be measurable. For example, the reproductive success rate of species of fish and growth rate of algae may be appropriate endpoints. It is critical to remember that an endpoint is an expression of the actual environmental value to be protected, such as spawning habitat for anadromous salmon. Management goals may be to increase reproductive success of salmon, maintain water quality, and increase spawning habitat. The important distinction between management goals and assessment endpoints, though, does not appear to be reflected in other studies and definitions (Suter, 1993).

Assessment endpoints are such important parts of the problem formulation phase of an ERA that poorly chosen or ill-defined endpoints can damage the credibility of any ERA. Ideally, endpoints should be measurable, but in the SLRA the ability to directly measure endpoints may not exist. Hence, extrapolation from existing data, perhaps on closely related taxa, is often used. Forbes *et al*. (2001) point out that most endpoints are based on studies using individual-level effects rather than population effects, and this focus can be misleading, as it assumes the individual fully represents the population. Another concern with extrapolation is that it ignores differences in life cycles of closely related taxa (Forbes *et al*., 2001). Barnthouse (1992) makes a point of differentiating between assessment endpoints (entity and it's attributes) and measured endpoints (e.g. toxicity tests, fields studies). He notes that to get to the assessment endpoint, one must extrapolate from the measured endpoint. The US EPA guidelines (1998) detail common problems associated with the selection of assessment endpoints.

The ERA process benefits from foreknowledge of management information needs and goals. As Pittinger *et al*. (1998) note: "Understanding the extent to which management options are known is valuable in helping to tailor empirical studies in order for the ultimate decisions to be made." In the context of the offshore oil and gas industry, government must supply guidelines for ERAs. Oil industry managers must make the purpose of the assessment known to the assessors. No level of management, however, should bias the process or be perceived to be biasing the process. Pettinger *et al*. (1998) warn that "[e]cological risk assessments should not be 'force-fitted' to justify preordained management decisions."

ECOLOGICAL RISK ASSESSMENT IN BRITISH COLUMBIA AND CANADA

The British Columbia government does provide guidance on ERA (Landis *et al*., 1998) and this process is built on the US EPA guidelines and those of the Canadian Council of Ministers of the Environment (CCME). The process has a highly guided, procedural structure in the form of a questionnaire that the assessor answers. Its current focus is entirely terrestrial and contaminant based. Questions are based on five main uses: industrial, commercial, residential, urban park, and agricultural. In a step-wise fashion, potential stressors and responses are determined. Risk characterisation is structured in a similar stepwise fashion, outlining in detail the level of calculations required. The guide even provides hypothetical calculations as examples.

The Environment Canada (EC) report *A Framework for Ecological Risk Assessment at Contaminated Sites in Canada*: *Review and Recommendations* (Gaudet, 1994) forms the basis of the EC approach to ERA. In concert with other ERA procedures, the report recommends a tiered approach to determine exposure assessment, receptor characterisation, hazard assessment and risk characterisation. The tiers involve an initial SLRA (Tier 1). Tier 2 is quantitative in model development and in data collection. Tier 3 involves increased use of site-specific data and predictive modelling capabilities.

The CCME comprises provincial, territorial and federal Ministers of the Environment. The aim of the council is to promote cooperation and coordination on interjurisdictional environmental issues. CCME releases, *A Framework for Ecological Risk Assessment: General Guidance* (1996) and *A Framework for Ecological Risk Assessment: Technical Appendices* (1997), as well as the EC report, use the tiered approach and are broadly similar to the US EPA guideline document(1998).

In the mid-1980s, the surge of interest in petroleum exploration off the west coast of British Columbia resulted in a plethora of reports concerning oil and gas exploration and the effects it could have on the marine environment. Chevron Canada (1982) and Petro-Canada Inc (1983) produced what was termed an Initial Environmental Evaluation (IEE) of the Offshore Queen Charlottes Islands. At this time, the concepts of ERA and ERM had not been developed. The reports could loosely be compared to a baseline EA, but the amount of data and knowledge used at the time was limited and incomplete. Some risk factors were considered such as the effect of operations, blowouts, environmental mitigations, and socio-economic concerns. A follow-up report from Chevron Canada (1985) dealt with a request for additional information from the West Coast Offshore Environmental Assessment Panel. The Chevron report considered effects on fisheries, physical, chemical and biological effects, socio-economic effects, physical environment effects, compensation issues, and risk scenarios.

In terms of present-day frameworks and guidelines, as set out by such agencies as EC, CCME, and the US EPA, there have been no formal, rigorous ERAs done for the west coast of British Columbia with respect to the potential stressors from an offshore oil and gas industry. As well, because it has been almost two decades since Chevron and Petro-Canada performed their IEEs, a fully developed EA would be required prior to, or concurrent with, the ERA process.

ECOLOGICAL MODELING AS A TOOL IN RISK ASSESSMENT

VALUE OF MODELS IN ECOLOGICAL RISK ASSESSMENTS

Models are developed to give credibility to ERAs. They can be quantitative, interpretative and predictive. In a broad sense, models fall into two categories: deterministic and probabilistic. Deterministic models in ERA are mathematical constructs in which the population and the environmental parameters in question are assumed to be constant and correctly identified (Suter, 1993). The deterministic models tend to deal with "worst-case exposure situations" (Solomon and Giddings, 2000) because they yield a single exposure value to compare with a single toxicity value. Probabilistic risk assessment (PRA) models attempt to quantify the relationship between likelihood of exposure and magnitude of effect. PRA models incorporate ecological concepts such as population dynamics, community resiliency and ecosystem function (Solomon and Giddings, 2000). They use the same information as deterministic models but they go beyond single-species single-dose toxicity data. PRA models are often the basis for extrapolation (See Suter, 1993 for discussion), but are not without criticism. As in deterministic models, information used in PRA models may not have much bearing on real-world exposures, for example, if exposure is pulsed and acute toxicity data are used when sublethal effects are critically important (for discussion see Scholz and Collier, 2000).

There are many kinds of models that range from individual species models to regional models that attempt to integrate temporal and spatial aspects of responses to stressors. It is important to note that models can be over-parameterised. Over-parameterised models often make data gaps appear irrelevant, when in-fact they are not, and "may convey a false impression of our understanding of risk" (Hill *et al*., 2000) and, depending on chosen parameter set, may lead to different management recommendations. The use of models, their methodology and their advantages/disadvantages are discussed in Suter (1993), and issues regarding the value of probabilistic models are discussed in Lee and Jones-Lee (1999), de Vlaming (2000), Woodburn (2000), and Solomon and Giddings (2000).

The Pacific Region of DFO has been fortunate that a number of modelling tools and experts are close at hand. William Crawford and Josef Cherniawsky of IOS ran the trajectory simulations of drift of surface-borne material presented in two accompanying Research Documents (Cretney *et al*., 2003; Crawford *et al*., 2003). In principle, these simulations could be run over and over again to encompass both likely and rare sea and air conditions. In this manner a probabilistic threat model could be generated; that is, the probability could be determined that oil released at any site would strike land at any other site under any possible meteorological and oceanographic conditions. Several ecosystem chemical fate models have been developed by Frank Gobas' group at Simon Fraser University and made available through the internet (www.rem.sfu.ca/toxicology/models.htm). A number of upper trophic-level models have been developed by researchers at the Fisheries Centre, University of B.C, and these too have been made available through the internet (http://www.ecopath.org/). SPILLSIM and OILMAP, which are 3D oil-behaviour models, have been licensed for use by DFO and/or DOE researchers. SPILLSIM was used to model simulated oil spills in several areas of the B.C coast (e.g., Hodgins, 1991; Hodgins, 1992). Major strides have been made in ecosystem structural and chemical-fate models in the last decade.

These models now may be regarded as reliable tools for environmental management. In a SETAC News article (1996), Don Mackay, the originator of multimedia environmental chemical fate models, states, "…managing the environment without a model is viewed by many managers (or perhaps potential mis-managers) as quite acceptable." He further states that "*…the model comes first, followed by experimental and monitoring tests of validity* [his Italics]." The models that Mackay refers to are mathematical, not conceptual models. These mathematical models produce quantitative results. In his article he observes, "Qualitative impressions of the state of the environment are often wrong, especially when the situation is complex with multiple chemical sources and pathways and subtle effects."

The comments of Don Mackay apply also to fisheries management for which sophisticated quantitative modelling tools are becoming available. Walters *et al*. (2000) comment that "[m]any plausible verbal explanations exist for the major abundance changes in some marine ecosystems…. There is a need to complement such 'word models' with the 'mechanized deductions' that models can do, so as to track all interactions at once, and hence avoid 'hidden assumptions' and logical inconsistencies that are all too easy to overlook in verbal explanation." Furthermore, Okey and Pauly (1999b) note that "[i]t is increasingly apparent that single-species approaches to fisheries management fail in all but the most fortuitous circumstances…."

Arguably, the single greatest advance made in ERA since the B.C. offshore oil and gas question was looked at in the mid 1980s is the development and refinement of ecosystem-level models. Many generic models can be downloaded free from web sites or are available for purchase. Almost all models can be run on a personal computer, after the necessary parameters and variable values for the ecosystem under study have been entered. Indeed, the ready availability of modelling tools is both a blessing and a curse. The model user must still have expertise in the field to understand the limitations of the models. Furthermore, good quality data are necessary to parameterise these models.

All mathematical models produce a facsimile of reality. Like art, the model result may be looked upon as abstract, cubist, impressionist, surreal, or realist. The result is never a photograph or moving picture show. Indeed, ecosystem models can be referred to as caricatures (Ruesink, 1999). "Caricatures are wrong but useful as long a they're not asked to perform tasks involving portions that are importantly wrong" (Ruesink, 1999). Thus, the user must have the expertise to understand the model and the ecosystem. The description of a model as a caricature is particularly apt, if you think about it. Prominent features tend to be exaggerated and others, diminished. In ecosystem trophic structural models, species less important to biomass and energy flow are left out or consolidated in some way. Unique or at-risk species likely may be left out because their inclusion will not alter the model output. Ecosystem, chemical-fate models generally are only applicable to chemicals chronically released into the environment. Accidental (acute) oil spill behaviour models are an exception.

Pharmacokinetic and pharmacodynamic models are developed to forecast the time course of chemicals in the tissues of organisms to evaluate their toxicological implications. However, such models "can only be an oversimplification of reality" (Wen *et al*., 1999). As Wen *et al*. (1999) note, predictive models built on theory may become invalid when the theory changes or is proved wrong. On the other hand, empirical models, which are built on observed relationships without theoretical underpinning, lack credibility when applied in unfamiliar situations.

The danger in mathematical models comes from an unquestioning reliance on them for management decisions. Schnute and Richards, (2001) discuss the role of quantitative mathematical models in fish stock assessment and the reasons for model failures. They note that "no fishery model can be completely trusted to capture biological reality." In view of the uncertainties in the capability of models to capture the true nature of systems, fisheries decision-makers have adopted a precautionary approach to fishery management (Richards and Maguire, 1998).

Model uncertainty may be best captured through use of Bayesian probability theory (Goodman, 2002) and decision analysis (Raiffa, 1968). Precautionary approaches, Bayesian statistics and decision analysis are discussed in more detail below.

TROPHIC-LEVEL AND CHEMICAL-FATE MASS BALANCE MODELLING

Modelling tools are available to forecast the fate and probable effects of toxic compounds distributed in an ecosystem from acute and chronic spills of oil and other chemicals from offshore oil and gas exploration, production, transportation and decommissioning activities. Chemical and oil spill models can be combined with food web models. Such model combining generally requires the interaction of disparate groups of modellers. Nevertheless, chemical fate models and ecosystem structure models require databases that have a portion in common. On this coast, the potential to achieve a high degree of collaboration among modellers is good. As noted above a group of chemical fate modellers resides at Simon Fraser University and groups of ecosystem structure modellers exist at the University of B.C. and within DFO. Indeed, the UBC group has developed higher trophic level, essentially steady-state (Ecopath), dynamic (Ecosim) and dynamic, spatially-segmented (Ecospace) models that are in widespread use. These UBC models are discussed in detail below, but it must be emphasised that other modelling groups and models exist with similar objectives.

TROPHIC-LEVEL, MASS-BALANCE MODELS

Upper trophic-level models

Ecopath with Ecosim (EwE) is an ecological software suite for personal computers that has been under development for more than a decade. The development is now centred at the University of British Columbia's Fishery Centre, while applications are widespread throughout the world (see http://www.ecopath.org). EwE has three main components: Ecopath - a static, mass-balanced snapshot of the system; Ecosim - a time dynamic simulation module for policy exploration; and Ecospace - a spatial and temporal dynamic module primarily designed for exploring impact and placement of protected areas.

The foundation of the EwE suite is an Ecopath model (Christensen and Pauly 1992, Pauly *et al.* 2000), which creates a static mass-balanced snapshot of the resources in an ecosystem and their interactions, represented by trophically linked biomass 'pools'. The biomass pools consist of a single species, or species groups representing ecological guilds. Pools may be further split into ontogenetic (juvenile/adult) groups that can then be linked together in Ecosim. Data inputs for Ecopath are generally already available from stock assessment, ecological studies, or the literature and include biomass estimates, total mortality estimates, consumption estimates, diet compositions, and fishery catches.

A preliminary Ecopath model for Hecate Strait included 27 species functional groups was developed by Beattie (1999). A recent version of the model has been expanded to include most of the Queen Charlotte Basin area and 53 species groups (Vasconcellos and Beattie, 2002).

Ecosim provides a dynamic simulation capability at the ecosystem level, with key initial parameters inherited from the base Ecopath model. Ecosim uses a system of differential equations that express biomass flux rates among pools as a function of time varying biomass and harvest rates, (for equations see Walters *et al.* 1997, 2000). The key computational aspects are:

- Use of mass-balance results (from Ecopath) for parameter estimation;
- Variable speed splitting enables efficient modelling of the dynamics of both 'fast' (phytoplankton) and 'slow' groups (whales);
- Effects of micro-scale behaviours on macro-scale rates: top-down vs. bottom-up control incorporated explicitly.
- Inclusion of biomass and size structure dynamics for key ecosystem groups, using a mix of differential and difference equations. As part of this assemblage, EwE incorporates:
  - juvenile size/age structure by monthly cohorts, density- and risk-dependent growth;
  - adult numbers, biomass, mean size accounting via delay-difference equations;
  - stock-recruitment relationship as 'emergent' property of competition/predation interactions of juveniles.
- Calibration of model forecasts with input time series data on relative abundance indices (e.g. surveys, catch per unit effort), absolute abundance estimates, catches, fleet effort, fishing rates, and total mortality estimates. Environmental "forcing functions", which are expressed as variations in primary production, may be estimated to optimise the fit between observations and predictions. These functions may then be compared with environmental time series in an attempt to identify possible forcing mechanisms.

For many of the groups, the time series data will be available from single species stock assessments. EwE thus builds on more traditional stock assessments, by using much of the information available from them, while integrating to the ecosystem level.

Ecosim provides two ways to explore impacts of alternative fishing policies:

- Fishing rates can be 'sketched' over time and results (catches, economic performance indicators, biomass changes) examined for each sketch. This approach uses Ecosim in a 'gaming' mode, where the aim is to encourage rapid exploration of options.
- Formal optimisation methods can be used to search for fishing policies that would maximise a particular policy goal or 'objective function' for management.

The goal function for policy optimisation is defined by the user in Ecosim, based on an evaluation of four weighted policy objectives:

- Maximise fisheries rent
- Maximise social benefits
- Maximise mandated rebuilding of species
- Maximise ecosystem structure or 'health'

Ecospace allows the evaluation of spatially explicit policies, such as the implementation of marine protected areas (MPA) on both large and relatively small scales (Walters *et al*., 1999). Ecospace is essentially a grid or matrix of 'cells', each cell consisting of a single Ecosim model (and initially identical as Ecosim inherits its parameters from Ecopath). This matrix of cells is then expressed at the user interface level as a map. Each cell in the map is linked through two processes: dispersal of organisms and redistribution of effort because of changing profitabilities and/or the creation of areas closed to fishing (Walters *et al*., 1999). The user creates the map, which is sketched in at the interface level using the computer mouse. Ecospace provides default settings that are easily changed at the interface level. The user can also sketch in patterns of relative fishing cost (effort is reduced or avoids high cost cells), patterns of relative primary production, and patterns of habitat to which species groups and fishing fleets can be assigned. A recent inclusion of certain Ecosim routines as components in Ecospace allow the estimation and comparison of the 'value' obtained from marine protected areas (MPAs), either as a function of the monetary value of fisheries or the gain/loss in biomass of groups/species.

EwE is an effective planning tool that requires explicit documentation of what is known and what knowledge is lacking for a specific system. Preliminary work in the Hecate Strait area has focused largely on upper trophic level commercial and recreational marine species (Beattie, 1999; Vasconcellos and Beattie, 2002). Whereas a considerable amount of information exists of diet, distribution, and abundance of the commercial species, less is known about marine mammals, sea birds, and forage fish. Information is being compiled on temporal variations in primary and secondary production, habitat distribution, and changes in the physical environment. These time series will be used to better understand trophic fluxes and to identify knowledge gaps in the Hecate Strait area.

A retrospective Ecopath model (Dalsgaard and Pauly, 1997) was developed for Prince William Sound (PWS), Alaska, which was the ecosystem into which the *Exxon Valdez* spilled its crude oil cargo in 1989. This model attempted to reconstruct the ecosystem as it existed between 1980 and 1985. Subsequently, an Ecopath model was built for the 1994-96 post-spill period and later refined with information gathered through the *Exxon Valdez* Oil Spill (EVOS) program. In 1999, the improved Ecopath model, together with simulations using Ecosim and Ecospace, was published as a comprehensive, UBC Fisheries Centre Research Report (Okey and Pauly, 1999a). Achieving the Ecopath model for PWS required collaboration among over 30 scientists and others to provide the essential input database (Okey and Pauly, 1999b). As the authors note, the Ecopath approach is flexible. New data can be easily incorporated in the models. The model results published in 1999 are not an end of the process, but merely a step stone leading to more refined models.

The Ecosim module was used to simulate a variety of catastrophic disturbances to the PWS ecosystem, including the *Exxon Valdez* oil spill. Okey, who wrote the section describing the results of the simulations, makes the following provocative assertions (Okey and Pauly, 1999a., p.80):

> "This analysis indicates that the working assumption that the Prince William Sound is recovering from the *Exxon Valdez* Oil Spill may be a reckless assumption. These simulations indicate that Prince William Sound, or other ecosystems, might not recover from disturbances that are severe across a broad range of trophic levels. Prince William Sound might 'stabilise' in an altered state."

The PWS ecosystem may stabilise in an altered state, but not necessarily because of the oil spill. Evidence is accumulating that climate change may be the driving force for ecological change in the Gulf of Alaska (Kaiser,1999). Thus, recovery cannot be assessed by comparing the current ecosystem of the sound to what it was before the oil spill. Rather, recovery must be assessed by comparing the current ecosystem to what it might have become due to climate change alone.

Even the notion of "recovery" requires definition. Recovery carries with it the anthropomorphic connotation of a return to the original state of health or "balance of nature." Ecologists are beginning to take a more complex view of ecosystems that departs from the idea that a "healthy" ecosystem is analogous to a healthy organism. Lancaster (2000) goes so far as to state, "*Ecological health* is a nebulous concept that should be expunged from the vocabulary. Likewise, all synonymous terms are ridiculous in a scientific context."

Landis poses the question: "If not recovery, then what?" as the title to a paper (2001). He then notes that the "question of what state should be the goal of restoration activities can not fall back upon the balance of nature as an effective model." In the same paper, he asserts that "[e]cological integrity and health are ambiguous terms that prevent a quantitative delineation of restorations goals". He proposes that there are "at least two ways of defining a restoration or recovery goal that are not dependent upon the failed paradigms of the past" (Landis, 2001). One way, he suggests, is "to define recovery goals based on a reference space derived from sampling so-called unimpacted sites." Another way, he suggests, is "to *a priori* describe an assessment space based upon the values of the stakeholders for the contaminated site." In either case a large number of variables may be used to describe "unimpacted" or "desirable" systems. The variables are likely to be highly correlated. For example, the biomass of certain species of zooplankton may be correlated with the biomass of certain species of phytoplankton that may be correlated with the composition and concentration of nutrient supply.

Fortunately, mathematical techniques are available to process data containing many variables by combining the correlated variables in such a way that a much smaller number of new variables are created that retain the information content of original variables. The number of these new variables defines the information dimension of the original data set. For example, the information dimension for 50 variables describing a given set of samples from unimpacted sites may be only three. If the new variables are orthogonal with each other, a 3D plot can be constructed in which the new variables form Cartesian axes and the samples form a cloud[3] of points. Samples from impacted sites can then be compared on the same axes to see if they fall within the cloud or outside of it. Given that all of the important variables have been included, if samples from an impacted site fall within the cloud, then restoration may be declared as achieved. If the information dimension is greater than three, then the samples for a hypercloud and comparison must be done mathematically rather than visually, but the process is the same.

Stakeholder-defined variables are measurement endpoints based on assessment endpoints. Given that consensus has been reached, the acceptable ranges of measurement values for the assessment endpoints define a hyperbox (Landis, 2001) in variable space. In the case of stakeholder-defined variables there are no sample data upon which multivariate techniques can be used to reduce the number of variables, although prior knowledge should be used to select useful assessment endpoints. Restoration is achieved when the impacted ecosystem moves within the hyperbox with time and cleanup effort (Landis, 2001).

Belovsky (2002) explicitly considers time in modelling the "cloud" that represents the acceptable or unacceptable states of a system. He points out that a system may not necessarily reach a single equilibrium state over time ("balance of nature") but may become a cloud of states, all of which may be considered "healthy." For example, ecosystems undergo a cycle of seasonal changes through time

To say that an ecosystem that is unperturbed by humans is healthy in one season and not in another requires a human value judgement. A human-caused perturbation that creates a winter ecosystem in summer, nevertheless, would be considered to have created an "unhealthy" ecosystem. Variations in state that are caused by the weather, nutrient upwelling, anthropogenic disturbances, and other conditions that are extrinsic to the system are called exogenous.

Variations in state may also be caused by endogenous conditions that are intrinsic to the system, such as predator prey interactions. Such interactions under the right conditions may lead to undampened oscillations in population numbers. The oscillations may be regular, such that the pattern repeats after a period of time, or completely irregular (chaotic). If a system is driven by natural conditions, the cloud formed by all states defined by the oscillations is "healthy", even though humans may value some states more than others. Even more complex systems can arise that result in more than one "healthy" cloud. Belovsky notes that "recent perspectives on the dynamics of ecological systems have identified the potential for systems to exhibit alternate states [a phenomenon which refers] to the possibility of two identical systems displaying different

[3] More than one cloud may form if the within-site variability is less than the among-sites variability. The within-site variability includes both the spatial and temporal variability (Landis, 2001). As long as the among-sites differences can be accounted for, restoration can be concluded if the sample cloud falls within the envelope encompassing the unimpacted sites.

baseline 'clouds' (e.g., interaction between predator and prey where the prey are limited by resources as well as predation and these processes occur at discrete time intervals…)."

The papers by Lancaster, Landis and Belovsky point out the complexity of ecosystems and the impossibility of defining the "health" of an ecosystem by analogy with the health of an organism. Lancaster and Landis reject the notion of "ecosystem health" or similar terms, whereas Belovsky, who is no less critical, does not seem prepared to reject the notion out of hand in his paper (2002).

An unexplored factor in the upper-level trophic simulations discussed above is that Ecopath and its linked models Ecosim and Ecospace do not include microbes in the model, even though, as noted by Ruesink (1999), the so-called "…'microbial loop' in pelagic systems can provide large portions of energy to higher trophic levels."

Lower trophic-level models

The possible importance of the lower-level, trophic structure of ecosystems to upper-level trophic structure has been underscored by recent simulations (Rivkin *et al*., 2000, 2001), in which a model pelagic ecosystem (Tian *et al*, 2000) was perturbed by nutrients from produced water. Produced water, which consists of formation and injection water[4], could exceed the total output of oil or gas by 10-fold over the lifetime of an oil field. These modellers observed that ammonia and dissolved organic carbon concentrations typically occurring in produced water from offshore oil and gas platforms had the potential to significantly increase pelagic concentrations of nutrients within 100 km^2 areas. When the ecosystem model was adapted for Newfoundland coastal waters and applied to the Hibernia site on the Grand Banks, it forecasted that the stimulation of added nutrients would lead to a change in ecosystem structure i.e., that small-celled phytoplankton and bacteria would be disproportionately favoured. As a result, Rivkin *et al*. (2001) concluded that "the fraction of PP [primary production] that is exported to mesozooplankton (and consequently to higher trophic levels) decreases, and the fraction vertically exported (and therefore transported to the benthos) increases." As a consequence, they felt that a change in the balance between pelagic and demersal fisheries could occur that favoured the latter. Furthermore, they expressed concern that the vertical flux of a greater proportion of PP could increase the benthic biological oxygen demand as well as sedimentary concentrations of contaminants, such as trace metals and hydrocarbons, from produced water. It should be noted, however, the results were preliminary. A full, peer-reviewed paper has yet to be published describing the simulations of nutrient addition through produced water.

Linking upper trophic-level and lower trophic-level models

The common practice is to develop upper or higher trophic-level (HTL) and lower trophic-level (LTL) models independently. To answer many questions, the HTL and LTL systems can be studied in isolation. For a more complete and truer model of reality, the systems must be coupled. The simplest method of coupling is to use the output of the LTL model as input to the HTL model. This approach may be useful for providing preliminary answers to 'what if' questions. Rivkin *et al*. (2001) used the potential effect of produced water in a LTL model system to infer effects on upper level organisms. These authors may have bolstered their case by using the LTL model results as input into an upper level model. Such a 'bottom-up' approach, however, implies that the HTL system has no or negligible influence on the LTL system. Nature is generally not so accommodating.

Discussions of the issues involved in coupling LTL and HTL models recently have taken place under the auspices of the PICES-GLOBEC Climate Change and Carrying Capacity Program (Rose, 2001; Eslinger *et al*., 2001; Zvalinsky, 2001; Ware *et al*., 2001; Rumsey, 2001; Kishi and Megrey, 2001). Rose (2001) emphasised the importance of temporal and spatial scales in coupling HTL and LTL models. He pointed out that the "processes controlling nitrogen, phytoplankton and zooplankton operate on different time and space scales than those important to fish, birds and mammals." He discussed the importance of feedbacks in coupling HTL and LTL models, especially negative feedbacks that stabilise population dynamics. Ware *et al*. (2001) outlined a number of other factors to consider in coupling of LTL and HTL models. They noted that alternative pathways should link lower to higher trophic levels, because "they allow LTL production to flow at different rates, to different HTL predators, under different environmental conditions." They identified copepods, euphausiids, and predatory zooplankton as "globally important LTL linking groups." They noted that most dynamic models have difficulty in handling more that ten HTL species (or functional groups) except

[4] Formation water co-occurs with oil in reservoirs. Injection water is injected into a formation to maintain system pressure and force oil to the surface.

for Ecosim, which can deal with up to fifty. They emphasised that number of species or functional groups included should be determined by the purpose of the model and that a variety of models may be needed to address species-specific questions. They suggested that likely models would be structured that had "a mixture of simple, pooled functional groups, and one or more size or age structured species groups." They considered the functional forms that the feeding rate of HTL predator on LTL prey could take and the influence of such factors as turbulent mixing, temperature as well as predator feeding and schooling behaviours. They pointed out that HTL growth rates are affected by size of food ration, metabolic rate and water temperature and that the optimum growth rate occurs at lower temperatures for smaller rations. They considered that this phenomenon had to be reproduced in the HTL component of a model. Finally, they acknowledged that seasonable migrations of HTL animals through subarctic marine ecosystems, as in the Queen Charlotte Basin, must be accounted for in models.

As noted above, an HTL, Ecosim model is under development for the Hecate Strait area of the Queen Charlotte basin. A complementary LTL model has yet to be chosen for the area. With the possibility of oil and gas development in the basin, the model of Tian (2000), as used by Rivkin *et al*. (2000, 2001) to investigate the potential effects of produced water on the Hibernia site of the Grand Banks of Newfoundland and Labrador, may be appropriate for adaptation to the west coast. On the other hand the Nemuro LTL model with the microbial loop included (Eslinger, 2001) may have practical advantages over the Tian's model. Future plans are to parameterise the Nemuro model to a coastal region and to link it with Ecosim under the GLOBEC-PICES umbrella through BASS (basin scale study) and, perhaps, REX (regional experiment) teams (Kishi and Megrey, 2001).

CHEMICAL-FATE, MASS-BALANCE MODELS

Multimedia equilibrium models (closed and open)

Chemical fate models come in a variety of complexities. Following Mackay (www.trentu.ca/academic/aminss/envmodel/), these models may be classified as Level I to Level IV with increasing complexity. The first three levels include models that are suited only to chemicals that are chronically released into the environment.

A Level I model simply partitions a chemical among compartments (e.g., air, aerosols, water, suspended sediments, biota, soil, and sediments) in a closed environment so that the chemical is at thermodynamic equilibrium. A "closed environment" is one from which the chemical cannot escape. At equilibrium, a chemical in each compartment has equal fugacity. Fugacity has the units of pressure and is formally related to it. Molecules of a chemical will tend to move from compartments of higher fugacity to those of lower fugacity. In a Level I model, transfer rates from one compartment to another are not considered, as a chemical is assumed to have instantaneously reached equilibrium (equal fugacity) in all compartments.

Level II models accommodate changes to the mass of a chemical in a compartment by its transformation to other chemicals or advection out of the ecosystem under study. A chemical is assumed to be released into and removed from the study environment (ecosystem) at the same steady rate. The chemical is assumed to be at thermodynamic equilibrium (has the same fugacity in all compartments), so that the rate of removal processes is smaller that the rate at which equilibrium is maintained. In this regard, it is worth considering the rates of removal of two chemicals that have the same half-life in air and water. If one chemical prefers to partition into a stagnant water body and the other, into a mobile air cover, the one preferring the air will be removed preferentially from the ecosystem, while the other will persist. Level II models allow for the calculation of chemical persistence.

Multimedia steady-state models

Level III models differ from level II models in that thermodynamic equilibrium in all compartments is <u>not</u> assumed for a chemical. Level III models are steady-state models, in which the input and output rates must be balanced, but the fugacities of chemicals can be at different levels in different media (air, water, soil, and sediment). In these models, transfer rates among the media can exceed the rates by which equilibrium is obtained. The tendency of the system to achieve a state in which a chemical has the same fugacity in all media is thus thwarted. The cumulative fate of compounds originating in many small oil spills over time can be modelled with a Level III model. Similarly, the fate of other chemicals lost to the environment through continual emissions from oil and gas operations can be modelled. Different compartments in the same media must have the same fugacity, however. Thus, the level III models do not accommodate food-chain biomagnification.

Multimedia dynamic models

Level IV models relax the requirements of equilibrium and steady state. These dynamic models can accommodate systems subject to single and discontinuous source inputs (Hertwich, 2001, and references cited therein). The Level IV models generally must be solved numerically, but can be solved analytically, when linear transport equations with constant transfer and decay coefficients can be used (Hertwich, 2001). Oil spill behaviour models that represent single large spill or blowout events perhaps may be considered as Level IV type models, although they are oleocentric. They model the fate of the oil and selected compounds that make up the oil, but do not generally model the fate of compounds in water, air and sediment. An exception is the recently developed multiphase (multimedia) oil spill model (MOSM) capable of modelling oil spill dynamics in 3D (Gin *et al*., 2001, and references cited therein). Neither Level III models, nor Level IV models encompass the biomagnification of chemicals in food webs.

Food-web bioaccumulation models

Food-web bioaccumulation models are required to forecast the fate of chemicals in organisms occupying higher trophic levels in food webs. Such models are becoming increasingly sophisticated and can, in principle, describe the fate of chemicals in a general food web that contains all relevant biota and encompasses air, water, soil and sediment (Sharpe and Mackay, 2000). A food-web bioaccumulation model suitable for hydrophobic organic chemicals in aquatic ecosystems is available from the modelling group at Simon Fraser University (http://www.rem.sfu.ca/toxicology/models.htm). Furthermore, the food-web models sometimes can be expressed in matrix form (Sharpe and Mackay, 2000). The ability to express the food web interactions in matrix forms brings the power of matrix algebra to the problem. For example, through matrix inversion, the proportion of a chemical that is contributed from each medium (air, water, soil, sediment) can be determined readily for all organisms in an ecosystem. Such a determination encapsulates chemical fate information in a format accessible to those who may not be familiar with the intricacies of food web chemical transfer. Indeed, Sharpe and Mackay (2000) state, "This format is ideal for conveying information to those outside of the scientific community, including policy-makers." The matrix food chain model is necessarily steady state, but can be adjusted to deal with growth dilution and failure to reach steady state (Sharpe and Mackay. 2000). As with the Level III models, the food-chain models deal best with chronic emissions. In the case of offshore oil and gas exploitation, they would be best suited to compounds being sporadically or periodically released over extended periods through drilling, production and transportation activities.

Dynamic oil-spill food-web bioaccumulation models

A dynamic oil spill-food web interaction model, which does not require steady-state conditions, has been built based on the multiphase oil spill model mentioned above (Gin *et al*., 2001). The marine food web consisted of trophic level, organism groupings (phytoplankton, benthic invertebrates, zooplankton, small fish, large fish), not single species, but the interactive model successfully demonstrates the coupling of oil spill and food web models. The coupled model was used to simulate an actual oil spill and satisfactorily forecast measured concentrations of the PAH anthracene in a species of benthic gastropod mollusc. No field measurements, though, were available for other organisms to test the model output. Only for the small fish did the model forecast concentrations of anthracene that rose to within an order of magnitude of lethal concentrations. Although anthracene, at a concentration of 5.6% in the spilled marine fuel oil, was the major aromatic compound, the model may have forecasted cumulative lethal toxic effect, had the phase concentrations of all aromatic compounds in the oil been computed.

CHEMICAL EXPOSURE MODELS

A variety of oil spill response and planning software tools (GNOME, ADIOS, TAP, DOGS) are available from the NOAA/Office of Response and Restoration (OR&R) facility in Seattle, Washington (http://www.response.restoration.noaa.gov/). The 2D trajectory analysis planning (TAP) method links release sites with receptor (impact) sites through "several hundred thousand test trajectories" (Galt and Payton, 1999). Databases are built up of the probability and degree of threat to each receptor site from any release site and chemical. These databases can be used with an environmental sensitivity database to assess risk.

Applied Science Associates (ASA), which supplies OILMAP, markets a spill impact assessment package (SIMAP). This package contains "a two dimensional oil trajectory model, a three-dimensional oil fates model, and a biological impact assessment model" (French *et al*., 1999). The package combines oil-behaviour process models for transport in and on the water, shoreline stranding and interaction, spreading, evaporation from slick, droplet entrainment, emulsification (water-in-oil), dissolution from slick and dispersed oil with partitioning to SPM, volatilisation from the water column, adhesion between SPM and oil droplets, sedimentation, mixing of sedimented oil in upper 10 cm of subtidal sediments, and photo- and biodegradation in all compartments. It provides the degree and extent of oiling of shorelines and concentrations of oil pseudocomponents in the water column. Risk to biota is determined through a biological effects submodel. The model can carry out probabilistic risk assessments. With the possible exception of the SINTEF (*S*tiftelsen for *IN*dustriell og *TE*knisk *F*orskning ved Norges Tekniske Hoegskole; *The Foundation for Scientific and Industrial Research at the Norwegian Institute of Technology)* 3D model system called OSCAR (*O*il *S*pill *C*ontingency and *R*esponse) (Reed *et al*., 1999), SIMAP is the most all-encompassing model available for oil spill risk assessment.

SIMAP, OSCAR and TAP are "exposure" models; that is, risk is determined by the exposure of organisms to oil and oil components. They do not include, as yet, modules covering uptake from water and food. Such capability is offered through the oil-spill food-chain interaction model by Gin *et al*. (2001). Another computer simulation software package for ERA that contains a food-web bioaccumulation submodel is EcoFate (Gobas *et al*., 1998; www.rem.sfu.ca/toxicology/). Although the package was not developed specifically for oil spills, it may have application to the cumulative toxicity of fugitive releases of oil and other chemicals from oil and gas operations. It has both steady-state and dynamic fate modules.

The DREAM (dose related effects assessment model) is under development to assess the impacts of all chemicals in produced water discharges into the marine environment (Karman, 2000). In this model internal, as well as external, concentrations of chemicals will be forecast for marine organisms. Furthermore, the model will include a time-to-event module for effects prediction. Inclusion of this module stems from recognition that in real-world systems exposure, and hence dosage, in organisms is time variable. DREAM will dynamically compute body burdens (residues) of chemicals and compare them to critical body burdens (residues) for chemicals based on experimental data or QSAR analyses. DREAM will assess risks summed over all components in produced water discharges and produce risk contour maps.

Finally, in the context of ERA, the risk of produced water to lower-level ecosystem structure (Rivkin *et al*., 2000, 2001) bears mentioning again. Also worth mentioning again are the upper-level trophic, mass balance models from UBC (Okey and Pauly, 1999a), which were applied in a retrospective assessment of the *Exxon Valdez* oil spill.

CAUTIONARY COMMENTS ON CHEMICAL AND TROPHIC MASS-BALANCE MODELS

The results from the ecosystem models may sometimes seem dire. Still, they are but warnings. The admonition by Ruesink (1999) that ecosystem models are caricatures of reality must be kept in mind. The model results tell what may happen, not what will happen. In this regard, it is worth reiterating Mackay's statement (1996) that "*…the model comes first, followed by experimental and monitoring tests of validity."*[ital. in original] Clearly, the models must be scrutinised, refined and tested. As well, the models must be adapted to B.C. marine ecosystems at risk from offshore oil and gas activities, before their results can have relevance to concerned parties and decision-makers.

MODELLING AND THE CONCEPTS OF INHERENT TOXICITY, CRITICAL BODY RESIDUE AND HORMESIS

INHERENT TOXICITY AND CRITICAL BODY RESIDUE

The concept of inherent toxicity is gaining broad-based support as a basis for defining the toxicity of substances (Gobas *et al*., 2001; Gobas, 2001; Mackay, 2001). The inherent toxicity of a substance is a measure of its internal concentration (internal dose). Gobas *et al*. (2001) suggests that a concentration of about 1 mmol/kg of body weight for any substance be accepted as the internal concentration that is associated with narcosis, the most elementary mode of toxic action. Substances for which there is evidence of toxic action below the internal concentration that would produce narcosis are to be regarded as being inherently toxic (Gobas *et al*., 2001).

Mackay (2001) suggests that Gobas *et al* (2001), in focussing on the internal concentration of a substance for the determination of inherent toxicity, make a "compelling statement of a Basic Principle relating to the nature of inherent toxicity of organic substances." A concentration of 1 mmol/kg of body weight may be regarded as the critical body residue (CBR) for non-polar narcosis (Mackay, 2001).

Mackay (2001) and McCarty and Mackay (1993) make a compelling argument for shifting aquatic toxicity evaluations from effects referenced to sediment or water concentrations to effects referenced to body concentrations. Toxicity determination then becomes a task of comparing body concentrations of chemicals to CBRs for different modes of toxic action. Mackay and McCarty (1993) note that "different modes of toxic action generally appear to be associated with differing ranges of body residues." Although their study shows some overlap among the ranges of body residue concentrations in fish for eight modes of toxic action and two exposure regimes, chronic and acute, the ranges involved are less than 2.5 orders of magnitude at worst. Some of the variability may be attributed to using body residue as a surrogate for the actual site of action. A toxicodynamic or pharmacokinetic model can provide a refinement for the time course of distribution within an organism, when necessary. Measuring whole body residues, rather than individual tissue residues, has cost advantages, especially for small fish. Mackay and McCarty (1993) suggest that the whole body residue approach can provide a common dose surrogate for those researchers doing population and community level studies and those doing physiological and biochemical studies.

The CBR approach offers a mechanism for dealing effectively with mixtures of chemicals (Mackay, 2001;McCarty and Mackay, 1993). This prospect should be attractive to those interested in the consequences of chronic and acute releases of oil and gas and other chemicals at sea. It should be possible to compute "the toxicity of single substances and mixtures from molecular structure" (Mackay, 2001)[emphasis in reference]. Oil is the quintessential mixture. Nonetheless, large groups of components, even if they are not analytically separable, have similar molecular structures and presumably similar toxic action. Similar compounds may be grouped to simplify assessments. Such a grouped mixture of components may be regarded collectively as a pseudocomponent (Stiver and Mackay, 1984) for some purposes. Moreover, some groups, such as the aliphatic and alicyclic hydrocarbons, probably do not exhibit inherent toxicity. A pseudocomponent simplification of data may be applicable to groups of oil constituents for fate modelling and determining collective exceedance of the level for narcosis.

Aromatic hydrocarbons are a group of chemicals known to have inherent toxicity. They may be evaluated together with other components by the pseudocomponent method to determine if the level for narcosis is exceeded by the total residual oil in organisms. Aromatic oil components likely will have to be handled as individual components for evaluation of inherent toxicity, such as photo-induced toxicity of polycyclic aromatic hydrocarbons (PAHs) (Gobas, 2001). McDonald and Chapman (2002) warn that the "ecological relevance of PAH phototoxicity remains uncertain; it should not be used for environmental management decisions unless its ecological significance is firmly established, and then only as a part of a weight of evidence determination."

Toxicity evaluation can be done on measured quantities of body residues of chemicals or on computed quantities of body residues from fate models. The latter method would be the one of choice in developing both pre- and post-spill response scenarios. It must be recognised, however, that carcinogenic aromatics constituents of the oil may "defy the definition of inherent toxicity as an inherent concentration threshold value may not exist (i.e., the single-hit hypothesis)" (Gobas, 2001). (For a somewhat contrary perspective, however, see the discussion on hormesis below.) Nevertheless, other modes of inherent toxicity for aromatic compounds can be modelled. Also, acceptable models of biodegradation and metabolism will be necessary to forecast fate and effects past the acute stages of an oil spill.

Researchers have taken up the challenge of testing and refining the CBR concept. As others have done recently, Lee *et al.* (2002b) found that CBR values for narcosis were time dependent. They observed a decrease in CBR of PAHs with an increase in exposure time in the amphipod *Hyalella azteca*. The toxicity was cumulative over time. These workers developed a damage assessment model that assumes that death happens when the tissue damage accumulates to a critical extent (Lee *et al.*, 2002a).

As a parting note to this discussion of the CBR, inherent toxicity approach, it is worth again quoting Mackay (2001):

"There is a need for a partnership in which results from carefully designed experiments to test the CBR concept are compared with predictions using QSARs, biouptake models and even physiologically based pharmacokinetic models. This is an exciting prospect that brings together the toxicologist, chemist, and modeller working to establish a scientifically sound basis for evaluating toxicity."

RELATIONSHIPS AMONG INHERENT TOXICITY, TISSUE QUALITY CRITERIA AND OTHER ENVIRONMENTAL QUALITY CRITERIA

Sediment and water quality guidelines relevant to B.C. offshore waters are developed for the Canadian Council of Ministers of the Environment (CCME, http://www.ccme.ca/publications/can_guidelines.html) and British Columbia Ministry of Water, Land and Air Protection (BC MWLAP, http://www.gov.bc.ca/wlap/). The guidelines are hazard based[5]. In contrast, the 56 substances on the Toxic Substances List in Schedule 1 of the Canadian Environmental Protection Act (CEPA) have been subjected to assessment of risk to human health. The actual risks of substances on the list to human health, however, still depend on case-by-case analyses of site and situation. The CCME guidelines also include tissue residue criteria that should not be confused with the CBR concept; the tissue residues refer to those of the diet, not those of the consumer. For the protection of marine organisms, the guidelines provide marine water quality criteria and marine sediment quality criteria for a limited number of constituents that may be found in crude oil or natural gas. Criteria are also provided in the guidelines for a number of other products that may be released during offshore oil and gas related activities. The list of products and criteria, however, is not as extensive as that provided in Patin (1999).

Water quality guidelines are generally well accepted, though they are not without drawbacks and controversy (Shepherd, 1999; Hart, 1999). Sediment quality guidelines, however, seem to be very controversial (O'Connor, 1999a,b; Noort *et al*., 1999; Fox, 1999; Long and MacDonald, 1999; Michelsen, 1999).

Shepherd (1999) discusses the case for deriving numerical tissue quality criteria (TQC) for the protection of aquatic life. CBR and TQC are burden-based, not exposure-based, criteria. The CBR concept seeks to relate body burden to threshold levels of modes of toxic action within an organism. TQC would be less directly related to internal modes of toxic action and would aim to protect at the community and population level, as the other guidelines seek to do. TQC can be related to biological effects through calculation from existing WQC or SQC. In principle, they may be derived from a database (e.g., www.wes.army.mil/el/dots/dots.html) of adverse effects and associated chemical residues in aquatic biota (Shepherd, 1999; Bridges *et al*., 1997). At present, there are no tissue quality guidelines "comparable to, for example, US EPA's ambient water quality criteria" (Shepherd, 1999).

As noted by Shepherd (1999), calculation of TQC for TQG purposes would be complicated by a number of factors. For example, what TQC can be set for PAHs in aquatic organisms capable of rapidly metabolising them? A surrogate of body burden, such as PAH adducts in DNA or metabolite conjugates in bile, may possibly be used. In addition, PAHs can be oxidised in the environment to give persistent, toxic products that can be bioaccumulated (Huang, *et al*., 1999). Still, in the case of bivalve molluscs, which have poor PAH-metabolising capability, studies of "*in-situ* sublethal physiological effects … have demonstrated clear relationships between biological impact and tissue residues of hydrocarbons derived from oil where this was the major contaminant" (Donkin, 1999).

As noted above, the CEPA Toxics chemical list contains only 56 substances for which human risk assessments have been done. Ross and Birnbaum (2003) promote a new direction in human and ecological risk assessment by advocating the integration of the two assessment processes (see also, US EPA, 1996) to provide a holistic assessment process. Integration is justified where the same mechanistic pathways operate in different species, and the modes of uptake are sufficiently similar. Ross and Birnbaum (2003) discuss aspects of an integrated assessment of persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs) and dibenzo-*p*-dioxins (PCDDs), in humans and wildlife. They note that a "major advantage of an integrated approach to risk assessment is the ability to gain mechanistic understanding that is based on multiple lines of evidence."

HORMESIS

Hormesis is a phenomenon that adds to the complexity of the ERA process. Hormesis is most often encountered in toxicity testing. In some cases, the measured response of a test group of organisms increases with decreasing dose (or decreases with decreasing dose) at dosages lower than the dose that provides a response not measurably different from that of the control group. Yet, at higher dosages in these cases the

[5] As noted by Chapman (2000), "generic EQVs [environmental quality values] only serve to identify hazard, not risk."

response is seen to increase with increasing dose (or decrease with increasing dose) in the expected manner for treatment with a toxicant. The dose response curve has a "U" or inverted "U" shape, depending on the response being measured. Although the term "hormesis" was coined in 1943, the phenomenon has been known since the late 19^{th} century (Calabrese and Baldwin, 2001).

Instances of the apparently often-beneficial effect of low dosages of petroleum on organisms can be found in the scientific literature. Reports of a stimulatory effect on aquatic plant growth or photosysnthesis from petroleum at low concentrations are common (e.g., Gordon and Prouse, 1973; Dunstan *et al*, 1975; Kauss and Hutchinson, 1975; Karydis, 1979; Morales-Loo and Goutx, 1990; McCann, *et al*, 2000). Hormetic effects of petroleum or its constituents have been claimed in the case of aquatic animals as well (Laughlin *et al*, 1981; Hose and Puffer, 1984).

Discussion of the phenomenon of hormesis has been confused by use of a variety of terms to describe it and similar phenomena (Calabrese and Baldwin, 2002a). The perceived connection and erroneous categorisation of hormesis with homeopathy (Calabrese and Baldwin, 2000; Chapman, 2002a) have added to the general suspicion about the reality of the phenomenon. To the detriment of the case for hormesis, this linking of the hormetic effects with homeopathy was made by the pharmacologist who is credited with showing in 1887 that the dose-reponse relationship could be biphasic[6] (Calabrese and Baldwin, 2001). Still, a recent in-depth, literature survey has supported the ubiquity and the validity of the concept (Calabrese *et al*, 1999; Holland, 1999). Such support does not allay suspicion about a phenomenon that allows for a low-concentration beneficial effect compared to controls of a toxicant that is not an essential element or vitamin.

To divorce the concept of hormesis from evaluations of harm and benefit, which are subjective assessments being dependent on point of view and circumstance, Calabrese and Baldwin (2002a) have provided an objective definition:

> "Hormesis is an adaptive response characterised by biphasic dose responses of generally similar quantitative features with respect to amplitude and range of the stimulatory response that are either directly induced or the result of compensatory biological processes following an initial disruption in homeostasis."

Calabrese and Baldwin's (2002a,b) attempt for a descriptive definition of hormesis has been generally well received by medical and environmental experts investigating hormesis (Chapman, 2002b; Carelli and Iavicoli, 2002; Kitchin, 2002; Pickrell and Oehme, 2002; Upton, 2002). The tone of the opinions is that the definition will provide a focus for future work and may be subject to refinement as more is learned about the phenomenon.

The definition specifies two types of hormesis: direct stimulation hormesis (DSH) and overcompensation stimulation hormesis (OCSH) (Calabrese and Baldwin, 2002a). DSH occurs as an immediate response of an organism to introduction of a stressor, whereas OCSH requires disruption of homeostasis, followed by a modest overcompensation and reestablishment of homeostasis again with the added feature of a biphasic dose response. The OCS hormetic response may exist for a finite period or indefinitely.

The amplitude of the hormetic stimulatory response over controls is less than two-fold and generally in the range of 30-60% over controls, and amplitudes greater than three- to four-fold probably represent a different phenomenon (Calabrese and Baldwin, 2002a). The dosage range for the hormetic effect tends to vary from one fifth to one hundredth of the toxic threshold, although ranges down to less than one thousandth have been reported (Calabrese and Baldwin, 2002a).

Distinguishing hormetic curvature from noise presents experimental difficulties when the amplitude of the departure from control response is expected to be two-fold or less. Adding to the problem in the case of OCSH is the requirement to repeat the experiment on treated organisms to determine if a biphasic response develops over time. These and other challenges in proving the occurrence of hormesis in an organism are discussed by Calabrese and Baldwin (2002a). A further complication also addressed by these authors is that no single underlying mechanism appears responsible for hormesis. On the other hand, they note that "a common evolutionary-based homeostasis maintenance regulatory strategy is evident."

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[6] In this context, biphasic means the dose-response relationship is essentially linear for doses greater than a certain threshold value and non-linear with a response minimum (U-shaped or J-shaped) or maximum (inverted-U or β-shaped) for doses less than that threshold value. The dose-response relationship has a linear phase and a non-linear phase.

The stimulated response may be deemed beneficial or not. That assessment can be made after an observation has been classified as hormesis. From the ERA and ERM perspectives the importance of hormesis is its probable effect at the community, population and ecosystem level. A net benefit to an organism of low dosages of a chemical or physical stressor may not translate into an overall benefit at higher levels of organisation. Chapman (2002a, 2002b) examines this issue in detail. He acknowledges that hormesis is "generally associated with single species adaptive responses" and advocates finding different terminology for "higher-level hormetic effects." Unlike some others, Chapman is not prepared to dismiss the importance of hormesis in ERA, but does warn that there is a need "to focus ERA efforts" and "not waste time or resources on trivial cases, which will solely denigrate hormesis as a general and acceptable scientific principle that is useful in ERA.

Because hormesis is a subtle effect evidenced at the individual and species level of organisation, ecosystem chemical fate and structural modelling may be the only way to assess the potential ecological impact of chemical stressors in the hormetic concentration ranges. Including hormesis in modelling increases the complexity of models and raises troubling questions. To begin with, is the phenomenon sufficiently understood and is the database extensive enough to attempt modelling? In the case of an oil spill, can the stimulation of photosynthesis and growth of phytoplankton in areas and times of low concentrations of oil offset the harm done in areas and times of high concentrations? Is the presumption tenable that nutrient supply is available to support increased phytoplankton growth? Would an increase in phytoplanktonic food for herbivores offset the toxicological harm that oil does to them? Do herbivores and other food-web organisms have their own hormetic range of oil concentrations?

Hormesis can no longer be dismissed in ERA and ERM. It seems clear that the scientific investigation of the phenomenon will accelerate in the future, given the interest in recent years. Calabrese and Baldwin (2003) assert that “[t]he hormetic dose response represents a paradigm shift in the concept of the dose response throughout biological science. It is widespread and outperforms other dose-response models. A general recognition of the hormetic perspective is likely to yield a vastly improved evolutionary basis of adaptive responses, scientific foundations of risk assessment and clinical medicine, as well as a more biologically plausible framework for understanding regulatory strategies at the level of the cell and the organism. Although regulatory and legal authorities have seldom been faced with the issue of hormesis, burgeoning research and past litigation will ensure that it cannot be ignored (Marchant, 2002).

Perhaps the most contentious issue regarding hormesis is the possibility that low doses of some carcinogens may reduce cancer risk (Calabrese and Baldwin, 1998a). "The implicit assumption in current risk assessment procedures, especially for carcinogenic substances, is that the dose-response function conforms to a linear, non-threshold (LNT) model." (Marchant, 2002). The dose-response function for hormesis is inherently non-linear at low dosages. Calabrese and Baldwin (1998b) have suggested that hormesis be considered as a default assumption in quantitative risk assessment, including carcinogen risk assessment. In this regard, it is worth noting the warning by Carelli and Iavicoli (2002) concerning hormesis that "any possible beneficial or harmful effects might lead to …misunderstanding…misinterpretation…or even cause political mismanagement of results." In addition, Marchant (2002) notes, perhaps prophetically, that "[h]ormesis is also likely to be controversial with the public, as the assertion that some of the most toxic pollutants known may be beneficial at low concentrations is likely to be met with incredulity and ridicule by many in the general public and media."

THE DECISION MAKING PROCESS

ECOLOGICAL RISK MANAGEMENT

Ecological risk assessment is an important tool in ecological risk management. Note that risk assessment and risk management, though linked, are separate activities. ERM is defined as "the process of identifying, evaluating, selecting, and implementing cost-effective, integrated actions that manage risks to environmental systems while emphasising scientific, social, economic, cultural, technological feasibility, political, and legal considerations" (Pittinger *et al*., 1998). Management decisions must be aimed towards achieving management goals. Risk management encompasses a wide range of largely subjective considerations that risk assessment, by its objective nature, does not. The overall goal of risk management is to minimise and manage exposure to hazards so that acceptable levels of risk are attained. The key phrase is “acceptable levels of risk” and the determination of acceptable levels is the risk managers primary task. The ERA process facilitates this task by presenting the risk manager with quantified probabilities and magnitudes of exposure and receptor response. The ERM process involves deciding among management options to select the "'most reasonable and acceptable option' under the given circumstances" (Pittinger *et al*., 1998).

Option selection includes consideration of the ecological and societal implications of each option. Concepts such as risk tolerance and perceived value (by the public, ecologists, and other stakeholders) must be taken into consideration. Pittinger *et al*. (1998) elaborate on a number of general ERM criteria garnered from participants at a workshop on ERM in 1997. Criteria relevant in Canada are:

- "Sound decisions should address the problem at hand without creating worse problems in other media or contexts.
- "Decisions should be meaningful and relevant to the risk.
- "Risk management should be innovative and flexible.
- "Benefits of risk reduction or avoidance should justify costs to society."

Pittinger *et al*. (1998) warn that "a decision must be made by risk managers without undue influence by a single stakeholder group" and warn against "highly compromised solutions reflecting an impractical marriage of all options." They also note that workshop participants broadly supported the proposition that decision-makers needed practical training in decision analytical techniques and risk management.

Risk management for marine resources has been frequently used for fisheries management and stock assessment issues. Francis and Shotton (1997) provided a comprehensive review of fisheries risk assessment and risk management issues. They reviewed the problems associated with variability and uncertainty, issues that are encountered during ERAs.

The US EPA (1998) also uses the concept of “risk assessors.” Risk assessors are individuals or organisations that bring expertise to the assessment team. They insure that scientific information is used effectively to address ecological concerns. The framework US EPA guidelines fully outline the roles of risk managers and risk assessors as well as what types of questions each group should ask.

There are many hurdles in the risk management process. ERAs are financially neutral in that they do not assess costs, but the risk management process involves making decisions about the cost-effectiveness of alternative actions or even the cost of no-action policies (Preston, 2000). Risk managers must also take into consideration society's values and concerns. This task can be especially difficult, as these values change with time. For example, in the 1950s wetlands were considered to have no societal value and were subsequently drained, primarily for agriculture. However, in the 1970s and 1980s, the public became educated about their value as critical habitat for wildlife, particularly waterfowl, and there was a subsequent movement to protect the remaining areas of wetland from agricultural and urban encroachment.

PRECAUTIONARY PRINCIPLE OR PRECAUTIONARY APPROACH OR BOTH?

In light of the uncertainties that a particular ERA may have, risk managers may wish to implement the 'precautionary principle'. The precautionary principle embraces the concept of 'prudent foresight.' When there may be a need to act even in the face of incomplete information, decisions should err on the side of caution to reduce the potential for unacceptable outcomes. The precautionary approach, which was initially developed by fisheries managers to mitigate unacceptable fisheries practices and problems with stock management, is equally applicable to all resource use and the making of decisions that could impact the environment. This concept is enshrined in Canada’s *Oceans Act.* The precautionary approach is necessary because it isn’t always possible to model, forecast, or mitigate the negative impacts of human activities (Hilborn *et al*., 2001). Recently, Canada’s Supreme Court recognised the use of the precautionary principle in their decisions in the *Chemlawn/Spraytech vs. Hudson* decision (Pelley, 2001). A brief history of the origin and use of the precautionary principle is given in a recent paper by deFur and Kaszuba (2002).

The precautionary principle has not been universally accepted and is the subject of debate (deFur and Kaszuba, 2002; Hileman, 2002). To some degree the debate has appeared to be one of semantics (Hileman, 2002). In the light of this debate, the expression "precautionary approach" may have been preferable to "precautionary principle" from the beginning. Nevertheless, the expression "precautionary principle" has become enshrined in policy and law. The Precautionary Principle Workgroup of SETAC (Society of Environmental Toxicology and Chemistry) has "concluded that the Precautionary Principle is a given…." (Sanderson, 2003).

A look in dictionaries gives flavour to the difference between the two terms. Among the relevant definitions, the Merriam-Webster Collegiate Dictionary defines "principle" as a "a comprehensive and fundamental law, doctrine, or assumption," "a rule or code of conduct" and "the laws or facts of nature underlying the working of an artificial device." Wordsmyth, the educational dictionary-thesaurus, provides these relevant definitions: " a law, doctrine, or assumption on which action is based" and "a law or rule that is presupposed or scientifically proven." In contrast, "approach" is simply "the taking of preliminary steps toward a particular purpose" and "a particular manner of taking such steps" by the first authority and "means or method of dealing with something" by the second. As it is used with "precautionary," "principle" implies something of substance, something that is comprehensive, universal and codified. Therein lies a problem with the term "precautionary principle." As yet, it lacks universal acceptance and meaning. Its formulation is therefore incomplete. Indeed, in January, 2002, John D. Graham, administrator of the White House Office of Information & Regulatory Affairs, is reported in a widely read trade magazine (Hileman, 2002) to have said that "the U.S. government supports precautionary approaches to risk management, but we do not recognize any universal precautionary principle." Graham's statement along with others on the precautionary principle and its meaning may be found in a dialogue report by Taylor and Ballantine (2002) summarising discourse at a January 2002 conference on *The U.S., Europe, Precaution and Risk Management: A Comparative Case Study Analysis of the Management of Risk in a Complex world.*

The 1998 Wingspread Statement on the Precautionary Principle (http://www.uml.edu/centers/LCSP/precaution/stat.wing.html) and the 2001 Lowell Statement on Science and the Precautionary Principle (http://www.uml.edu/centers/LCSP/precaution/stat.summ.html) have wide support and are, perhaps, the best articulations of the principle to date. The former statement provides an expression of the precautionary principle: "When an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically." The latter statement reaffirms the former and sets forth the following elements for its effective implementation:

- "Upholding the basic right of each individual (and future generations) to a healthy, life-sustaining environment as called for in the United Nations Declaration on Human Rights;
- "Action on early warnings, when there is credible evidence that harm is occurring or likely to occur, even if the exact nature and magnitude of the harm are not fully understood;
- "Identification, evaluation and implementation of the safest feasible approaches to meeting social needs;
- "Placing responsibility on originators of potentially dangerous activities to thoroughly study and minimise risks, and to evaluate and choose the safest alternatives to meet a particular need, with independent review; and
- "Application of transparent and inclusive decision-making processes that increase the participation of all stakeholders and communities, particularly those potentially affected by a policy choice."

Regarding the implication of the precautionary principle, a communication on the precautionary principle from the Commission of the European Communities (2000) highlights the following points, among others:

- "Recourse to the precautionary principle presupposes:
  - identification of potentially negative effects resulting from a phenomenon, product or process;
  - a scientific evaluation of the risk which because of the insufficiency of the data, their inconclusive or imprecise nature, makes it impossible to determine with sufficient certainty the risk in question.
- "The implementation of an approach based on the precautionary principle should start with a scientific evaluation, as complete as possible, and where possible, identifying at each stage the degree of scientific uncertainty.
- "An assessment of the potential consequences of inaction and of the uncertainties of the scientific evaluation should be considered by decision-makers when determining whether to trigger action based on the precautionary principle. All interested parties should be involved to the fullest extent possible in the study of various risk management options that may be envisaged once the results of the scientific evaluation and/or risk assessment are available and the procedure be as transparent as possible.
- "Measures should be proportional to the desired level of protection
- "Measures should not be discriminatory in their application
- "Measures should be consistent with the measures already adopted in similar circumstances or using similar approaches.
- "The measures adopted presuppose examination of the benefits and costs of action and lack of action. This examination should include an economic cost/benefit analysis when this is appropriate and feasible.

However, other analysis methods, such as those concerning efficacy and the socio-economic impact of the various options, may also be relevant. Besides the decision-maker may, in certain circumstances, by guided by non-economic considerations such as the protection of health.

- "Measures, although provisional, shall be maintained as long as the scientific data remain incomplete, imprecise or inconclusive and as long as the risk is considered too high to be imposed on society. Maintenance of the measures depends on the development of scientific knowledge, in the light of which they should be reevaluated. This means that scientific research shall be continued with a view to obtaining more complete data. Measures based on the precautionary principle shall be reexamined and if necessary modified depending on the results of the scientific research and the follow up of their impact.
- "Measures based on the precautionary principle may assign responsibility for producing the scientific evidence necessary for a comprehensive risk evaluation."

The above points make it clear that the Commission of the European Communities regards application of the precautionary principle is part of risk management. The points make clear as well that the precautionary principle should only be applied after a thorough scientific evaluation that includes the identifying the degree of uncertainty. The Commission's communication also requires that a risk assessment be done, whenever possible, before applying the precautionary principle and describes in an Annex the four components of risk assessment that must be done: hazard identification, hazard characterisation, appraisal of exposure, and risk characterisation.

While the communication from the Commission of the European Communities provides useful clarification of how and when the precautionary principle should be applied, it is essentially silent on the question of "precautionary approach" versus "precautionary principle" that is posed in this section. On this question, Garcia (1996) makes distinctions among four basic fisheries management approaches that Auster (2001) has generalised to habitat management. These approaches are: preventative approach, corrective approach, precautionary approach and precautionary principle. Auster classifies these approaches along two axes: level of uncertainty and potential cost of errors (ecological and economic). The preventative approach applies when uncertainty is low, although costs may vary from low to high. The corrective approach applies when potential costs are low, although uncertainty may vary from low to high. The precautionary principle applies when uncertainty is high and costs of errors may include irreversible damage. The precautionary approach applies for in-between situations.

A discussion document of particular importance to Canada is *A Canadian Perspective on the Precautionary Approach/Principle* (http://www.ncr.dfo.ca/cppa/menu.htm). The document consolidates the perspectives of 12 departments and agencies as well as the Privy Council Office and the Treasury Board Secretariat. Although the document recognises that "…there may be distinctions to be drawn between them, the terms 'precautionary approach' and precautionary principle' are used interchangeably…." Still the term "precautionary approach" is preferred throughout the document, and the statement of the elements of implementation is couched in that phraseology. The document specifically affirms that application of the precautionary approach should be science based. It recognises "concerns that it [precautionary approach] could be applied to perceived risks for which there is no sound scientific basis" and provides guidelines that seek to assuage that and other concerns. The discussion document places the precautionary approach "as a decision-making tool within risk management.

The publications reviewed above accept that the precautionary principle is applied after a risk assessment, if one can be done, or are silent on the subject. A contrary view held by some (e.g., Santillo and Johnson, 1999) is that the precautionary principle should displace risk assessment. Santillo and Johnson (1999) assert that "the precautionary principle is a higher-order paradigm that guides the decision-making process from problem identification through to action, and not simply a management tool to be invoked when a risk assessment identifies substantial residual uncertainties." They discuss flaws that can affect risk assessments and management decisions. These flaws include overlooking or wrongly excluding scenarios that may turn out to be important and relying on point estimates without appreciating their uncertainties. Santillo and Johnson (1999) feel that the precautionary principle "engenders the aspiration to achieve a progressive reduction in environmental burden, without a reliance on the need to identify and quantify specific risks." Needless to say, the views of Santillo and Johnson (1999), and the like minded, are disputed by others. Fairbrother and Bennett (1999) "argue that the Precautionary Principle remains a management philosophy, not a substitute for risk assessment." Pittinger and Bishop (1999) note that "[p]roponents of what might be considered to be more extremely cautious interpretations of the precautionary principle (Santillo *et al.*, 1998) have argued that current risk-based assessment approaches are overly narrow or under-protective, and should be substituted with more preventative hazard-based approaches" and argue that "[w]hile hazard assessment is a fundamental, necessary element of risk assessment, we do not view it as sufficient or pragmatic basis on which to make risk management decisions involving products or their formulation."

Most of various discussion documents mentioned above place precautionary approaches, including the precautionary principle, as tools within risk management. This distinction is important because some people have considered that precautionary approaches are tools within risk assessment. Problems arise from the use of worst-case or default parameter values in risk assessment, as a hedge against underestimating possible hazard and exposure because of scientific uncertainty. Uncertainty factors, which do provide an element of precaution in risk assessment, may be used without proper documentation (Bonnell and Atkinson, 1999). Then, use of the precautionary approach by an incompletely informed risk manager may lead to double accounting of uncertainty. Because precaution is so deeply integrated into quantitative risk assessment (Hileman, 2002), some overlapping use may be difficult to avoid. A way out of this dilemma may be found through the use of Bayesian statistics and decision analysis.

BAYESIAN STATISTICS

Only Bayesian statistical analysis presents a theoretically justifiable means of providing the probability that a hypothesis (analyte concentration, parameter value, ecosystem model, ERA etc.) is true, given the experimental evidence. The probability that something is true is what scientists, regulators, assessors, activists and the 'man in the street' would like to know. Unsaid in this assertion about Bayesian statistics is the corollary: classical, frequentist statistics do not provide such probabilities. At best all that classical statistics can tell us is how well evidence collected fits a hypothesis, given that the hypothesis is true. Classical statistics has its place, but the results tend to be grossly misinterpreted. Innumerable papers through the years have made this point, but the interpretative sins abound to the present.

The distinction may be made clear from an examination of Bayes theorem:

$$p(H_i \mid \mathrm{D}) = \frac{p(H_i) \times p(\mathrm{D} \mid H_i)}{\sum_{i=1}^{n} p(H_i) \times p(\mathrm{D} \mid H_i)}$$

where $p(Hi|\mathrm{D})$ is the probability that hypothesis “i” is true, given the data, $p(Hi)$ is the prior probability that hypothesis “i” is true, $p(\mathrm{D}|Hi)$ is the probability of the data, given hypothesis “i” is true, and the sum in the denominator is the total probability of the data and ensures that $\Sigma\, p(Hi|\mathrm{D}) = 1$. Frequentist statistics deals with $p(\mathrm{D}|Hi)$. In the case, where contaminant concentrations are being measured, $p(\mathrm{D}|Hi)$ provides the distribution or scatter of the measured concentration values about the mean value.

Kaiser (1970) states: "The distribution function (of any type), gained by a finite series of repeated measurements, gives the average $\bar{x}$ of measurements and information about their scatter around this average, but it does not say how near this particular average may be to the 'true value,' which is supposed to be equal to the mean $\bar{X}$ of the whole population if no systematic errors, causing bias, are present. …. A different function is needed which states how the possible true values are distributed around one given measured value." That function ($p(Hi|\mathrm{D})$ is provided by application of Bayes theorem. Kelly (1972) has provided a simple example of using the Bayesian approach to provide the distribution of true values about the measured value of a chemical element in a powder. Wade (2000) provides a useful tutorial on Bayesian methods in conservation biology.

As stated here, if the Bayesian approach is superior to the frequentist or "classical" approach, then the question arises: "Why isn't everyone a Bayesian?" (Efron, 1986). As Efron (1986) points out, everyone used to be Bayesian up to the 20th century, and modern statisticians have provided powerful theoretical support for Bayesian inference while cataloguing the inconsistencies of the frequentist perspective. The answer to the question, as provided by Efron (1986), is that frequentist statistics are easy to use compared to Bayesian statistics, pay close attention to experimental design, allow pieces of a complex problem to be calved off for solution, and have seized the high ground of objectivity. This last claim about objectivity, however, is disputed (see below). To Efron's reasons may be added the "absence from the undergraduate curriculum" of Bayesian statistics and general cautiousness of corporations, government regulatory agencies, and academic institutions (Malakoff, 1999).

An argument may be made that the vehicle for the resurgence of Bayesian statistics has been the increasingly powerful and inexpensive desktop computer. As noted by one statistician in Malakoff's article (1999) when referring to past applications of Bayesian methods, "[i]t might take just a half hour to write down the equation but forever to do the computation." Along with desktop computer power have come new software tools that simplify the application of Bayesian methods (Malakoff, 1999; Schnute *et al*, 1998). A software program known as BUGS (Bayesian inference using Gibbs sampling) that is built on the MCMC (Markov chain Monte Carlo) simulation method is available for free ((http://www.mrc-bsu.cam.ac.uk/bugs/). It has been used in a wide range of applications, such as cost-effectiveness analysis, fisheries stock abundance modelling, pharmacokinetic modelling, actuarial modelling and epidemiological modelling.

Lack of objectivity has been seen as the 'Achilles heel' of Bayes statistics in what is known as the prior or *a priori* distribution, which incorporates advance estimates of the probability that competing hypotheses (parameters, models, ERAs, etc.) are true. Because little advance information may be available to choose among them, the probabilities over the range of hypotheses may be chosen as equal. Such a prior distribution is considered as non-informative, although it may be possible to specify upper and lower bounds. Experts may be able to provide a non-uniform ranking of hypotheses, but their opinion may be biased, too narrow or too wide.

An examination of the issues of the failings and misinterpretations of frequentist statistics, the pitfalls of expert opinion, the advantages of Bayesian approaches to decision-making in the context of ERA is provided by Germano (1999). On the issue of the objectivity of frequentist statistics, Germano notes, paraphrasing noteworthy statisticians, that "classical statistics are just as subjective as Bayesian analysis, and that subjectivity is expressed in the choice of significance level and the choice of the statistical model used for the final analysis (which can have a much greater impact on the outcome than the choice of a prior distribution)." Despite the 'I may be subjective, but so are you too' argument, much work has gone into the formulation of informative priors in Bayesian analysis, since the mid-1980s. Much of this work has been done in the application of Bayesian analysis to fisheries stock assessment and management.

In the light of the title of this document, it is reassuring that scientists are studying the properties of Bayesian methodology in dealing with the uncertainties in stock assessment models (e.g., Fried and Hilborn, 1988; Walters and Ludwig, 1994; Cox-Rogers, 1997; Adkison, 1998; Sainsbury, 1998; McAllister and Kirkwood, 1998; Hill, 1998; Schnute, 1997; Rivot *et al*, 2001; Nielson and Lewy, 2002). Thus, Bayesian models in ERAs of offshore oil and gas activities will not be regarded, one hopes, with unwarranted scepticism. A pioneer in the application of Bayesian statistics to fisheries problems, Hilborn (1999) (http://www.pnl.gov/statenvi/ssenews/ENVR_3_1.pdf) states the essence of the value of Bayesian statistics in ERA with the words:

> "The most important thing for a statistician to understand in fisheries risk analysis is that the role of statistics is to calculate the probabilities of alternative hypotheses for use in decision analysis. The end product of a risk analysis in fisheries, and I assume all environmental risk analysis, is a decision, and any statistical tools that help refine our understanding of the consequences of alternative decisions are useful. Tools that don't provide an aid to decision making are not useful. Thus I have been drawn by pragmatism to Bayesian statistics simply because it appears to be the only branch of statistics that ever claims to be able to calculate the probability of different hypotheses being true. Decision makers are not interested in the distribution of the estimators, but rather in the distribution of the real world."

Bayesian and frequentist statistics are in a struggle for ascendancy. In 1975 Lindley forecast a Bayesian 21st Century (1975). In 1986, Efron (1986) checked the balance sheet and concluded that the "practical advantages of Fisherian/frequentist methods …so far seem to have outweighed the philosophical superiority of Bayesianism." In 1994, from the perspective of the decision analysis field Ward Edwards, then Chair of the Decision Analysis Society of INFORMS (Institute for Operations Research and Management Science), argued that "the 21st Century will be the Century of Bayes" to which he added "QED[7]" (Edwards, 1994) (http://decision-analysis.society.informs.org).

Forecasts aside, undoubtedly the Bayesian and frequentist schools will coexist for some time to come (Malakoff, 1999). The adherents to the two schools can be expected to keep each other in line - a good thing, if done without rancorous debate. Scientists who use frequentist statistics will no longer get away with

[7] quod erat demonstrandum, Latin for: which was to be demonstrated or proven

misinterpreting the meaning of their results. Scientists who use Bayesian statistics will be watched for unjustified priors.

Risk managers and other decision-makers ultimately may choose the ascendant school in this century. Being presented with probabilities of competing hypotheses may turn them in favour of Bayesian statistics. Frequentist statistics cannot deliver such probabilities

DECISION ANALYSIS

Decision analysis is an adjunct process to decision making that enables the practitioner to decompose complex problems into both tractable and uncertain parts and then logically recompose the parts into models that demand coherence (consistency) in thought from decision-makers. As noted for other models, decision analysts' models approximate reality. All rational outcomes are included. Although not all possible outcomes of decisions may be encompassed, analysts hope that the really important ones are. Some outcomes may remain unforeseen and others may be considered baseless. A decision analyst investigating the ramifications of introducing freon to the market place would not have foreseen that the refrigerant would cause an ozone hole over Antarctica. Postulating such an effect at the time would likely have been considered baseless speculation anyway. A far-seeing expert may have foreseen the possibility, though, in which case the event could have been included with an estimate of probability.

Decision-analysis models incorporate a number of characteristics useful to decision-makers. The models explicitly include uncertainty. They can assess how sensitive outcomes are to uncertainty in specific parts of the model. The models permit incorporation of expert advice on relevant parts of a problem; i.e., they allow, indeed they encourage, a team approach. Expert advice must be incorporated with caution, though (Germano, 1999; Peterman and Anderson, 1999). Decision analysis models can include a quantitative expression of the risk tolerance of a decision-maker. In addition, the models provide a logical record of the decision process.

The origins of decision analysis can be traced to World War II when groups of scientists of disparate disciplines, such as physics, biology, statistics and mathematics, were organised into teams to tackle critical problems (Raiffa, 1968). After the war, the business community began to embrace systems analysis, of which decision analysis is a branch, and a host of related analytical processes that evolved from the war-time team activities that became a discipline known as 'Operations Analysis' or 'Operational Analysis' (Raiffa, 1968).

The basic tool of the decision analyst for complex problems is the decision tree or decision-flow diagram. It is the framework on which is built an analysis model. As noted by Bunn (1984), "[t]o many people, decision analysis is the decision tree model." The decision tree consists of sequences of decisions and outcomes. Decisions are controlled by the decision-maker; outcomes are controlled by chance. In the tree, by convention, a square represents a decision node (or branch point) and a circle, a chance node (Raiffa, 1968). Certain rules apply to tree growth. For example, branches cannot coalesce. All possible decisions and outcomes must be represented (Bunn, 1984). Indeed, a chance node may have an infinite number of branches growing out of it (but only one growing into it). However, even with only two branches growing from each node in a perfectly symmetric tree, the number of branches at each growth stage increases exponentially (as 2^n), as do the number of scenarios represented by different paths from the stem along the branches to the tips. Needless to say, with the number of scenarios potentially increasing exponentially, a complex model can be computer intensive. Selective pruning according to strict criteria and arrested branching may help keep a tree to a manageable size.

Uncertainty is incorporated into decision tree models through the chance nodes. For instance, at some stage in a decision tree model, it may be necessary to use a mass balance model, such as those discussed above. A variety of competing models may be available each having a different uncertain view of how nature operates and including a variable number of parameters of uncertain value. In addition, the models will be subject to "intrinsically unpredictable (random) components of the future events" (Goodman, 2002). In principle, because all possible decisions and outcomes should be represented, scenarios should be developed for each model, parameter value and random component. To keep the models computationally tractable and as uncomplicated as possible continuous functions may be discretised (Bunn, 1984) or their contributions obtained by integration ((Bunn, 1984; Walters and Ludwig, 1994). Analytical integration may be possible, but numerical integration techniques may be required (Walters and Ludwig, 1994). Computerised simulation methods are available for generating a probability distribution around functions of parameters. These methods are particularly useful in sensitivity analysis, which is used to determine how dependent decision model outcomes are to variations in parameter and probability values. Decision-makers then can know how robust the rank order of decision options is to uncertainties in a decision analysis (Peterman and Anderson, 1999).

Peterman and Anderson (1999) provide a modern, reasoned discussion of decision analysis in the context of ERM. They examine how resource managers have dealt with uncertainties in the past and point out the advantage of decision analysis, in which handling uncertainty is an intrinsic part of the process. They note with elaboration that decision analysis separates a complex problem into eight components: "(1) management objectives, (2) management options, (3) uncertain states of nature, or uncertainties yet to be resolved, (4) probabilities on the uncertain states of nature, (5) model to calculate the outcome of each management option for each state of nature, (6) decision tree, (7) ranked management options, and (8) sensitivity analysis." Except for management objectives the other components are directly incorporated within the decision tree framework. Management objectives help specify management options and set criteria for ranking management options. As note by Peterman and Anderson (1999), however, "many resource management agencies find it remarkably difficult to define a specific, clear, and unambiguous objective that can be used as a criterion to rank order their various management options." Peterman and Anderson (1999) provide candid discussions of challenges to implementing decision analysis and communication its results and well as its benefits. In this regard, Raiffa (1968) and Bunn (1984) in their books also provide illuminating and candid discussions of the 'pros and cons' of decision analysis. The final section in the article by Peterman and Anderson (1999) provides a synopsis of a number of cases in which decision analysis was used successfully in ERM.

An aspect of decision analysis that has only been touched on is that most decision analysts now appear to be Bayesians rather than frequentists. Raiffa, considered to be a founder of decision analysis, admitted to being a Bayesian, when it was the minority group (Raiffa, 1968). Decision analysis seems to be in a worldwide growth phase that Bayesian decision analysts are leading (Edwards, 1994).

REGULATORY OVERVIEW

Both provincial and federal governments in Canada have various commissions and regulatory processes (outside of EA and ERAs) that must be addressed. These features of regulatory overview are provided in a review by Jacques Whitford Environmental Limited (2001). The review includes discussions on the role and activities of the B.C. Oil and Gas Commission, the B.C. Environmental Assessment Process, the role of public consultations and the federal government's Environmental Impact Assessment Process. The background report of the Marine Awards Society of Canada (Johnston and Hildebrand, eds.,2001; http://web.uvic.ca/masc/pubs.html) provides a useful summary of the jurisdiction and roles of the various agencies that have an interest in the Canadian offshore.

The British Columbia Scientific Review Panel provided comment on the regulatory environment needed for a future B.C offshore oil and gas industry in their two volume report of Jan. 15, 2002 (Strong *et al*., 2002a; Strong *et al*., 2002b; (http://www.em.gov.bc.ca/Oil&gas/offshore/default.htm). The report recommended that "…a quantitative risk analysis be undertaken as a vehicle for decision-making by the various stakeholders." The report noted that "…several sectors of natural resource development and management have been marked in recent years by divisions among specialists on the weight that should be given to precautionary considerations in situations where there is credible risk of substantial environmental damage arising from any project under consideration." In addition, the report recommended that "…the BC government might wish to consider setting up an arms-length mechanism (e.g. through the province's educational institutions) that would both provide the general public with periodic summaries or abstracts of the technical literature, written in non-technical language, and also receive, interpret and communicate data from local and independent observers." This arms-length mechanism might provide a suitable way to deal in a non-inflammatory manner with controversial issues, such as the precautionary principle, hormesis, and Bayesian statistics.

SUMMARY OF KNOWLEDGE AND KNOWLEDGE GAPS

EA and ERA are similar but fundamentally different processes. The latter process involves the specification and quantification of risks. ERA and ERM are now recognised as interdependent, though distinct, functions in dealing with ecological risk. They have accepted definitions with published framework guidelines. Still, there is room for refinement.

A three-tiered approach to ERA appears to have become the norm. The tiers increase with their number in detail and complexity. Tier-one is synonymous with screening-level ERA (SLRA), whereas the other tiers are detailed-level ERAs (DLRAs). Specifying appropriate assessment and measurement endpoints in ERA separate from management goals continues to be a challenge in ERA. Variability and uncertainty are vital characteristics of ERAs that are essential to decision-makers. An unknown degree of uncertainty and variability exists due to multiple stressors in ecosystems, when single-stressor data are generally all that is available

Mathematical models give credibility to ERAs and ERM decisions, although they can be misused and mistaken. Probabilistic models appear to have advantages over single-scenario deterministic models, in part because the variability and uncertainty is 'built in.' Probabilistic models provide more information than deterministic models to decision-makers and may lead to different choices among management options. Models aid, but do not substitute for, expertise. They simulate reality and do not encompass it.

A preliminary upper-level trophic, mass-balance Ecopath model is available for the Hecate Strait area of the Queen Charlotte basin. A dynamic Ecosim model is under development for the same area. Ideally, a spatially resolved Ecospace model would be desirable to have for the whole Queen Charlotte basin. It would be desirable as well to have a coupled lower and upper trophic level model for the basin. In addition, it would be nice to have an ecosystem, chemical-fate model built on this coupled model.

A number of chemical exposure models are available that have been developed for, or are adaptable to, the offshore oil and gas industry. These chemical exposure models are useful in quantitative ERAs for acute and chronic releases of oil and other chemicals from the industry. Such models may be parameterised for areas in the Queen Charlotte basin. Exposure modelling, however, requires a firm knowledge of what organisms there are, where they are, and when they are in the ecosystems under the possible threat from the oil and gas industry.

Incipient knowledge gaps are developing now that will need addressing in the perhaps decadal role-out to the establishment of an oil and gas industry off the B.C coast. These incipient gaps stem from emerging issues, such as a shift from exposure models to body residue models in toxicology, a shift from linear, no-threshold and linear, threshold dose response models to non-linear, hormetic dose-response models, and a shift from frequentist to Bayesian statistics in ERA.

Environmental systems are complex. Decision analysis is a logical process developed to deal with complex systems. Decision analysis can ensure coherence, consistency and transparency in the decision-making process. Still, decision analysis is but an aid to the decision-maker.

RECOMMENDATIONS

- A guidance document should be prepared that specifies the tier-levels and timing of ecological risk assessments that governments will request from offshore oil and gas industry proponents
- Government scientists should carry out in-depth reviews of chemical-fate and trophic-level, mass balance models available, with a view to specifying the performance requirements (suitable level of complexity, track record, etc.) of models used by the offshore oil and gas industry in developing ERAs. In this regard, some models currently developed or under development for coastal B.C. by academic and government scientists may be useful for ERA purposes.
- In view of the apparent movement of research on chemical fate and toxicological studies towards the concepts of body residue and inherent toxicity, a decision to move away from the present practice of estimating risk based on exposure may be prudent for the future. Such a decision will have to be made to provide guidance to industry for the preparation of ERAs.
- Guidance on the application of hormesis in ERA will require an in-depth review of possible ramifications of the phenomenon, some of which may be extremely controversial. Despite the possible ramifications, the decision must be science-based not policy-based.
- Some clarification is required in the debate surrounding the terms "precautionary principle" and "precautionary approach."
- Guidance is needed on whether or not Bayesian statistics should be a requirement of ERAs produced by proponents of the B.C. offshore oil and gas industry.
- Guidance is needed on whether or not decision analysis should be required as an adjunct to decision making.
- A program of public education should be a key component of initiatives to deal with controversial issues

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