IMPLEMENTING SCIENCE IN ENVIRONMENTAL ASSESSMENT— A REVIEW OF THEORY

by

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ABSTRACT

Despite a growing body of literature addressing the scientific requirements of competent environmental assessment (EA), practice remains contested. This study aims first to provide an overview of scientific developments associated with EA since the 1970s, as evidenced in the peer-reviewed literature. The second objective is to judge, on the basis only of evidence in the peer-reviewed literature, whether scientific theory and practice are at their vanguard in EA and related applications. Through the review, I reflect on biophysical science as it applies to stages of the EA process. I also reflect on debates surrounding the role of science as it relates to political and administrative dimensions of EA. Based on this review, I am convinced that science inside EA has not kept pace with developments in science outside EA. I also believe that improvements to the quality of science in EA will rely on the adoption of stronger participatory and collaborative working arrangements.

LIST OF ABBREVIATIONS USED

AEAM	Adaptive environmental assessment and management
AVHRR	Advanced very high resolution radiometer
BACI	Before-after-control-impact
CCCEAC	Committee on Climate Change and Environmental Assessment in Canada
CEA	Cumulative effects assessment
CEAA	Canadian Environmental Assessment Agency
CEQ	Council on Environmental Quality
EA	Environmental assessment
EIA	Environmental impact assessment
EIS	Environmental impact statement
ESSA	Environmental and Social Systems Analysts Ltd
ERL	Environmental Resources Ltd
GIS	Geographic information system
GPS	Global Positioning System
IBP	International Biological Program
IGBP	International Geosphere Biosphere Program
LIDAR	Light imaging, detection, and ranging
LTER	Long Term Ecological Research Network
MEA	Millennium Ecosystem Assessment
MODIS	Moderate resolution imaging spectroradiometer
NAS	National Academy of Sciences
NASA	National Aeronautics and Space Administration
NGO	Non-governmental organization
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
NUSAP	Numerical, unit, spread, assessment, pedigree
PNS	Post-normal science
SA	Sustainability assessment
SEA	Strategic environmental assessment

SFM	Sustainable forest management
UNCED	United Nations Conference on Environment and Development
UNFCCC	United Nations Framework Convention on Climate Change
VEC	Valued ecosystem component
VHF	Very high frequency
WCED	World Commission on Environment and Development
WCRP	World Climate Research Program

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CHAPTER 1 INTRODUCTION

1.1 Problem and Context

Formal environmental assessment (EA) processes were established by governments in North America during the early 1970s to satisfy growing public concerns over the environmental impacts of uncontrolled industrial and economic development. This marked the beginning of a vast new enterprise of environmental decision-making, complete with regulatory requirements, scientific contributions, and participatory processes. Today, EA everywhere appears to be in a state of crisis, as scholars and practitioners alike continue to find the entire process seriously wanting (e.g., Ross et al. 2006). While some (e.g., Doelle 2012; Gibson 2012) criticize governments for ineffective administrative processes, others (e.g., Doelle and Sinclair 2006; Stewart and Sinclair 2007) find much to be improved upon in participatory mechanisms. Still others (e.g., Fairweather 1994; Treweek 1995; Warnken and Buckley 1998; Greig and Duinker 2011) find major fault in the scientific dimension of EA. The situation observed today is puzzling, since the basic principles of competent EA were published decades ago (e.g., Sadler 1996). Indeed, strong scientific principles for EA were established in the early literature of the 1970s and 1980s (e.g., Holling 1978; Beanlands and Duinker 1983). Despite a growing body of literature outlining the basic requirements of competent EA, it appears the EA community continues to struggle with undertaking useful and defensible scientific investigations in EA practice.

What was the original purpose of EA? During the 1970s, the language used in EA laws and policies was that of environmental protection, i.e., the protection of important environmental values. To this end, administrative bodies were established by governments to translate new EA laws and policies into regulatory procedures and requirements. The results of these early implementation efforts, however, were promptly challenged by members of the scientific community (e.g., Andrews 1973; Carpenter 1976; Schindler 1976) who observed that a founding principle of EA—science in the service of environmental decision-making—was not finding application in EA practice. Indeed, many scientists observed that existing procedural frameworks for EA had favoured the production of comprehensive but superficial environmental descriptions, rather than incisive environmental impact predictions. It was concluded, therefore, that EA was not providing decision-makers with the critical knowledge of impacts needed to achieve EA's original aim: the protection of important environmental values.

Recognizing the need to operationalize EA's scientific requirement, government agencies, universities, and NGOs in North America mobilized prominent research scientists to develop rigorous investigative frameworks for conducting EAs (e.g., Sharma et al. 1976; Andrews et al. 1977; Holling 1978; Ward 1978; Munn 1979). This early guidance—collectively inspired by the interdisciplinary science of ecology—established foundational principles and protocols for predicting the biophysical impacts of development alternatives.

During the 1980s, researchers and practitioners continued to elaborate strong scientific principles and protocols for conducting EAs (e.g., Fritz et al. 1980; Duinker and Baskerville 1986; NRC 1986a; Walters 1986; Duinker 1989). Despite these ongoing conceptual and technical developments in the literature, the overall adequacy of EA practice remained widely contested (e.g., Rosenberg et al. 1981; Clark et al. 1983; Bisset 1984; Hollick 1986; Culhane et al. 1987; Fairweather 1989). In response to ongoing weaknesses in EA practice, government agencies in North America funded major initiatives—one American (Caldwell et al. 1982) and one Canadian (Beanlands and Duinker 1983)—to investigate possibilities for improving the scientific basis for EA. While both reports identified some important technical advancements in EA practice since the early 1970s, they concluded that ongoing science challenges in EA could be largely attributed to organizational and political factors. In general, both reports observed that existing review formats had fostered adversarial relations between those responsible for preparing EAs (proponents and consultants) and those responsible for reviewing them (EA administrators, legal professionals, research scientists, public interest groups, and other intervenors). This had created an atmosphere of conflicting expectations, widespread confusion, and general frustration within the EA community. Consequently, both reports observed the need to move away from adversarial review formats to focus on building more collaborative arrangements for EA design and preparation.

In the Canadian project's final report, Beanlands and Duinker (1983) outlined a series of so-called requirements for conducting ecological studies in EA. Most notably, they proposed the early identification of valued ecosystem components (VECs) in order to guide subsequent research activities. The VEC concept would go on to become a widely influential device for focusing assessments on stakeholder-relevant issues. To avoid the problem of costly and frustrating adversarial reviews, Beanlands and Duinker (1983) also proposed the establishment of a two-stage scientific review process. This process would rely on close collaboration between scientists, proponents, and consultants in developing mutually agreeable research designs prior to their implementation and final review. Overall, I consider the Beanlands and Duinker (1983) report, together with the literature of the 1970s, to provide the basic foundations for designing and implementing rigorous scientific investigations in EA practice.

By the early 1990s, it was clear that the ultimate purpose of EA had been broadly recast in terms of sustainable development (WCED 1987; Sadler 1996). In a major international review of EA practice, Sadler (1996) observed the need to reconcile political, scientific, and administrative expectations within the EA community to foster a more coherent and unified process for securing sustainable development decisions. In the project's final report, Sadler (1996) targeted four stages of the generic EA process for immediate and cost-effective improvements: scoping, significance evaluation, document reviews, and follow-up monitoring. Consistent with Sadler's (1996) recommendations, researchers and practitioners of EA have continued to elaborate strong approaches related to scoping (e.g., Mulvihill and Jacobs 1998; Mulvihill and Baker 2001; Mulvihill 2003), scenario-building (e.g., Duinker and Greig 2007), research collaboration (e.g., Greig and Duinker 2011), significance evaluation (e.g., Lawrence 2007a, 2007b, 2007c), effects monitoring (e.g., Arts 1998; Morrison-Saunders and Arts 2004), and biodiversity assessment (e.g., Nelson and Serafin 1991; Gontier et al. 2006).

Despite these ongoing developments in the literature, a considerable gap appears to persist between proposed ideals and their practical implementation. I maintain that if the global EA enterprise is to fulfill its ultimate purpose—to contribute to a sustainable pattern of development by protecting VECs—researchers and practitioners around the world must strive to adopt a collaborative, participatory, and scientifically rigorous approach to conducting EAs.

1.2 Objectives

This review aims first to provide a major summary and synthesis of scientific developments associated with EA since the early 1970s, as evidenced in the peer-reviewed formal literature. In addition to key scientific developments, the review considers important political, organizational, and administrative factors related to the implementation of science in EA. The second objective is to judge, on the basis only of evidence in the peer-reviewed literature, whether scientific theory and practice are at their vanguard in EA-related applications.

1.3 Scope

The review focuses broadly on empirical and deductive biophysical science as it applies to the generic EA process. Key topics addressed by the review include: the role of science in EA; scientific foundations of EA; conceptual frameworks for conducting science in EA (e.g., adaptive management, post-normal science, and transdisciplinary imagination); emerging ecological concepts for EA (e.g., resilience, climate change, and landscape ecology); and key methods, tools, and techniques as they relate to stages of the generic EA process (i.e., scoping, baseline characterization, impact prediction, significance evaluation, decision-making, and follow-up). The review also explores important administrative and political factors as they relate to implementing useful and defensible scientific investigations in EA. While the review draws heavily on literature addressing the North American EA experience, generalizations are made wherever possible in order to make overall lessons more applicable to the worldwide EA enterprise

1.4 Outline

Following a brief outline of methods, the paper begins with some theoretical constructs for delineating the role of science in EA. Since EA is about more than just science, it seems important to begin by examining how science in EA can help to foster sustainable and mutually-agreeable development decisions. After situating science within the broader EA enterprise, I move on to explore the scientific foundations of EA. I then examine four conceptual frameworks for undertaking scientific investigations in the face of conflict and uncertainty. From here I transition into emerging ecological concepts for EA. Following explorations of these higher-order themes, I treat important principles, protocols, and techniques as they apply to each stage of the generic EA process. The paper concludes with a synthesis of major scientific developments associated with EA since the early 1970s, highlighting important challenges and opportunities for practical implementation.

CHAPTER 2 METHODS

In today's parlance, I conducted a comprehensive or narrative overview of the scholarly literature surrounding EA-related science. As such, the review is both descriptive, in summarizing the contents of the items retrieved, and critical, in judging the quality or validity of the findings expressed therein.

With the help of my supervisory committee, I built a hierarchical set or organizing themes and concepts to structure the review. Based on several decades of experience as practitioners and scholars of EA, my committee was able to help me identify and group several key concepts that have shaped collective thinking about EA-related science since the 1970s. At the finest level of detail, we identified particular kinds of tools and techniques (e.g., remote sensing, simulation modeling) that might be used at different stages of the EA process (e.g., baseline characterization, impact prediction). At a much broader level, we identified several conceptual frameworks (e.g., adaptive management, post-normal science) that have shaped contemporary thinking about the implementation of science in EA. We also identified a number of ecological concepts (e.g., biodiversity, climate change) that have begun to find their way into EA-related scientific applications over the last few decades. Lastly, we decided that a review of the EA-related scientific literature would not be complete without consideration for EA's other two dimensions (i.e., political, administrative), so we included them as minor themes in the review as well.

I began obtaining relevant literature by conducting searches in three major electronic databases: Google Scholar, Web of Science, and Scopus. Database queries typically comprised specific scientific terms (e.g., indicator, simulation model) combined with generic EA terms (e.g., environmental impact assessment). Such phrases were carefully constructed to query all three databases exhaustively, thereby ensuring comprehensive coverage of the literature. To ensure even coverage through time, I consistently sorted all search results by decade (e.g., 1970s, 1980s). In addition to searching electronic databases, I conducted hand searches of EA textbooks, government reports, as well as official and semi-official guidance materials. As a final means of ensuring complete coverage, I consulted the literature-cited lists of all the authoritative articles and books retrieved through earlier searches. This last step in the retrieval process yielded a considerable volume of relevant material.

In my attempt to capture a wide range of literature, I deliberately adopted a broad definition of what constitutes EA-related science. Because science in EA spans many academic disciplines, I was forced to draw on literature from diverse scientific fields situated outside EA, particularly the science of ecology. Moreover, where the literature offered important political and administrative perspectives on implementing science in EA, I made sure to include them in the review.

Literature items obtained through searches (~5,000) were first organized into a hierarchical set of file folders based on important concepts and search terms (e.g., biodiversity, scoping, prediction). Each item was then carefully examined by me to determine its relevance to the review. As part of the vetting process, I constructed a spreadsheet database arraying relevant literature items (~800) against their contents based on keywords (e.g., climate change, resilience, prediction). In the end, my comprehensive approach to canvassing combined with my highly selective approach to screening ensured that far more literature was initially retrieved than was included in the final review.

Though I am aware that important advancements in EA-related science may also be found in the regulatory filings associated with formal EA practice, such grey literature was deemed outside the scope of this review. I do note, however, that many of the scholarly articles reviewed here did draw heavily upon such grey literature. It is my hope, therefore, that certain elements of EA practice have made their way, if only indirectly, into this review.

CHAPTER 3 CONCEIVING A ROLE FOR SCIENCE IN EA

3.1 Science and Politics

Formal EA processes around the world require various combinations of scientific investigation and public participation to support the ultimate goals of environmental protection and sustainable development. However, because there is often little guidance on how to integrate scientific and political elements in practice, a divergence of expectations has emerged within the EA community (Sadler 1996). In a recent survey of EA practitioners, Morrison-Saunders and Sadler (2010) found that while most respondents recognized the fundamental need for both scientific and political inputs to EA decision-making—especially with respect to achieving sustainable development decisions—they could not necessarily explain how science and politics should be integrated in practice.

Indeed, there are multiple ways to place science and politics into a theoretical construction of EA (e.g., Bartlett and Kurian 1999; Cashmore 2004). According to Cashmore (2004), the role of science in EA can be seen along a spectrum ranging from EA as applied science to EA as civic science, with five distinct models identified within the two paradigms. In the applied science models (analytical science and environmental design), the methods of the natural sciences prevail. In the civic-science models (information provision, participation, and environmental governance), the emphasis is on stakeholder participation and value judgements. According to Cashmore (2004), the perceived role of science in EA depends largely on one's views regarding the immediate purpose of EA, i.e., how it interacts with decision-making processes (informing, influencing, or integrating). Cashmore (2004) argues that because science in EA has merely sought to interact with decision-makers through passive information provision, it has largely failed to shape the final outcomes of their decisions. Moreover, Cashmore (2004) suggests that demands for stronger participatory politics in EA—aimed at actively influencing development decisions—have arisen from a growing awareness of science's failure to sway decision-makers. In short, Cashmore (2004) places science and politics at odds with one another in the battle to influence sustainable development decisions.

I prefer to understand the relationship between science and politics in EA within the context of Lee's (1993) framework for pursuing sustainable development. In his landmark 1993 book entitled *Compass and Gyroscope: Integrating Science and Politics for the Environment*, Lee (1993) compellingly argues that a sustainable pattern of development is most effectively pursued through the integration of experimental science and participatory politics. To this end, Lee (1993) outlines a process of collective social learning which relies on adaptive management (Holling 1978; Walters 1986) to reduce scientific uncertainty, and principled negotiation to bound stakeholder conflict. For Lee (1993), adaptive management and principled negotiation together serve as complementary navigational aids (i.e., "compass and gyroscope") in the pursuit of sustainable development. In the context of Lee's (1993) framework, EA is conceived as an ongoing process of structured social learning in which experimental science, oriented by bounded stakeholder conflict, can provide useful and defensible knowledge to stakeholders and decision-makers in the collective pursuit of sustainable development. In other words, the immediate purpose of EA—understood here as the creation and mobilization of reliable knowledge in support of sustainable development—is most effectively achieved through the fusion of adaptive management and principled negotiation.

3.2 Science Inside and Science Outside

Around the world, independent reviews of EA findings are typically required in cases of controversy and conflict. In such cases, independent review bodies may call upon research scientists to provide expert testimony during adversarial review hearings. While this sort of arrangement does allow research scientists to participate in the evaluation of EA findings, it does not encourage them to collaborate with consultants and proponents during the design and implementation of scientific studies.

Greig and Duinker (2011) prefer to place research scientists into more constructive roles with respect to EA practice. According to Greig and Duinker (2011), science outside EA (e.g., Ducrotoy and Elliott 1997; Lester et al. 2010) is needed to create robust ecological effects knowledge for practitioners, while science inside EA (e.g., Beanlands and Duinker 1983) is needed to apply that knowledge to generate reliable impact predictions for decision-makers. Greig and Duinker (2011) observe that because professional consultants working inside EA are often faced with severe time and resource constraints, research scientists outside EA are needed to assist in creating reliable ecological effects knowledge over longer time frames. Indeed, research scientists operating outside EA have the capacity to mount long-term process studies in large ecosystems, undertake experimental cause-effect research for specific development types and VECs, and assemble general effects knowledge into models for predicting the impacts of development alternatives. To reciprocate, science inside EA is able to provide practical applications and specific test cases for the refinement of general effects knowledge generated by science outside EA.

To enable the sorts of organizational arrangements needed to support collaborative impact research, Greig and Duinker (2011) propose the establishment of multi-stakeholder research networks resembling those that have supported similar sustainable development initiatives (e.g., SFM Network 2016). Research networks intended for EA would comprise unique partnerships between universities, governments, businesses, and NGOs according to specific development types, VECs, and regional contexts. These coordinated research networks would engage EA practitioners in designing the targeted research programs needed to generate robust ecological effects knowledge, thereby contributing to sustainable development while reducing scientific uncertainty over time.

The position taken by Greig and Duinker (2011)—that the EA community continues to struggle with scientific practice—stands in contrast with the findings of Morrison-Saunders and Bailey (2003) and Morrison-Saunders and Sadler (2010), who surveyed EA practitioners on two occasions and found that they were in large measure pleased with the quality of science in EA, but displeased with the importance placed on science at various stages of the EA process. Greig and Duinker (2011) responded by observing that such findings are not surprising, since the science practiced inside EA is under the direct purview of those very practitioners. Greig and Duinker (2011) also pointed out that EA practitioners would obviously defend the quality of their own work, and would naturally express disappointment when others did not place much importance on it. In short, Greig and Duinker (2011) maintain that substantial improvements can and should be made to the quality of science practiced inside EA. To do so, however, will require the establishment of collaborative research networks linking scientific practice inside EA with scientific research conducted outside EA.

Barriers to achieving such collaborative partnerships are well-known (e.g., Briggs 2006; Rogers 2006; Roux et al. 2006; Gibbons et al. 2008). According to Ryder et al. (2010), such obstacles arise from the fundamentally different perceptions that scientists, managers, and stakeholders have of science and scientific knowledge. To address this, several environmental laws and regulations have sought to standardize scientific contributions to resource and environmental decision-making by requiring that such decisions be based on so-called 'best available science' (e.g., Bisbal 2002; Glicksman 2008; Mills et al. 2009; Green and Garmestani 2012; Hanekamp and Bergkamp 2016; Murphy and Weiland 2016). Oddly, such mandates have failed to include an explicit definition of what constitutes best available science, or how it might be applied to environmental decision-making.

Attempts to elaborate on the concept (e.g., Bisbal 2002; Sullivan et al. 2006; Ryder et al. 2010; Green and Garmestani 2012) have observed that non-scientists tend to view science as a collection of information, whereas scientists themselves tend to see science as a systematic process of gathering and organizing knowledge into theories and testable hypotheses. Such ambiguity has led to disagreement over the precice meaning of the term 'best available science'. Proposing a way forward, Ryder et al. (2010) identified several core attributes for best available science. Ryder et al. (2010) also observed that these principles are clearly embedded in the science of adapative management. In the words of Ryder et al. (2010): "The creation of interdisciplinary teams in an adaptive management framework is an essential process to identify research questions, create a 'taxonomy' of available information and facilitate the incorporation of new scientific information as it becomes available". It is clear, then, that the creation of the best scientific information inside EA will rely not only on the application of the best scientific tools and techniques available, but on the adoption of collaborative working arrangements that facilitate productive researcher-pactitioner relationships.

3.3 Science, Politics, and Administration

Sinclair et al. (2017) write that all environment and resource decision-making can be broadly conceptualized in terms of three fundamental dimensions: (i) the administrative/regulatory dimension, which is dominated by government responsibilities, timelines, and required procedures to obtain development approval; (ii) the participatory/political dimension, which is dominated by stakeholder relations, conflict, power, and civic engagement; and (iii) the scientific/technical dimension, which is dominated by scientific protocols for developing and testing impact predictions. The scientific dimension (including both traditional and local ecological knowledge) aims to deliver the critical understanding of potential impacts that informs discussions and debates in the other two dimensions. In the context of Sinclair et al.'s (2017) framework, all three dimensions require satisfactory implementation if the central purpose of EA—now commonly agreed upon as sustainable development—is to be achieved. Avoidance of the administrative/regulatory dimension may mean that strong results from scientific and participatory processes have no influence on government regulators. Poor performance in the participatory/political dimension may mean that robust scientific results within a strong regulatory framework go unheeded as communities rise up and oppose development. Weak science, even in situations of strong participatory processes and successful navigation of bureaucratic administration, may lead to developments that have undesirable environmental impacts.

In sum, science—conceived here as the production of reliable knowledge about the potential biophysical impacts of development alternatives—is an absolutely necessary yet wholly insufficient element of competent EA. Indeed, formal EA processes around the world, as defined by the regulatory requirements under which most EA is conducted, typically demand a scientific contribution in the form of baseline characterizations, cause-effect research, impact predictions, and monitoring programs. Clearly, EA cannot be conducted as a strictly participatory or governance-oriented process. Of course, neither can it be conducted as a strictly scientific process, for "influence without insight is dangerous, but insight without influence is useless" (2015 personal communication with Peter Duinker).

CHAPTER 4 FOUNDATIONS OF SCIENCE IN EA

4.1 Early EA Methods

Early methods for conducting EA have been comprehensively reviewed elsewhere in the literature (e.g., Bisset 1978; Canter 1983; Shopley and Fuggle 1984; Wathern 1984). These methods are generally divided into four broad categories:

(i) Overlays
(ii) Matrices
(iii) Checklists
(iv) Networks

The use of spatial overlays in EA was first suggested by landscape architects at the University of Pennsylvania (McHarg 1969). Using this technique, analysts would visualize the spatial implications of alternative transportation routes by overlaying a series of shaded or coloured transparencies depicting important environmental factors with transparencies depicting development characteristics. In the 1970s, following the expansion of geographic information systems (GIS), rasterized data layers could be used in place of transparencies to quantitatively depict environmental factors and development characteristics. Using this approach, analysts assign numerical values to each raster-cell in a given data layer, overlay multiple data layers, and then aggregate the overlapping numbers to obtain spatially explicit estimates of impact magnitude/significance.

The most well-known of the early EA methods is undoubtedly the Leopold matrix, developed and published by the US Geological Survey (Leopold et al. 1971). This large rectangular array lists 100 generic development actions on the horizontal axis against 88 generic environmental components on the vertical axis, for a total of 8,800 potential interactions. By mentally "overlaying" project proposals onto site descriptions, analysts assign separate magnitude and significance ratings (1-10) to all potentially relevant interactions. These dimensionless ratings may then be summed to obtain a total impact score.

Weighted-checklist methods for EA include the Environmental Evaluation System, developed by Battelle Laboratories for the US Bureau of Land Reclamation (Dee et al. 1973). This method employs expert-derived value functions and weights to transform 78 estimated water quality parameters into 18 dimensionless water quality scores, each representing the "condition" of a generic environmental component. In this framework, an impact is defined as the difference between estimates of current (unaffected) and future (affected) environmental conditions.

The use of causal network diagrams in EA was first suggested by a student at the University of California (Sorensen 1971). Using so-called Sorensen networks, analysts draw linkages between development actions and linear sequences of environmental components (typically two or three of these). They then qualitatively describe expected development-induced changes in those components. The original Sorensen networks were intended to depict relationships linking generic development actions (associated with one of 55 pre-defined coastal development types) to qualitative changes in six categories of generic environmental components.

4.2 Challenges from the Scientific Community

Objections to early methods—particularly matrices and checklists—can be found throughout the early EA literature (e.g., Andrews 1973; Lapping 1975; Bisset 1978; Holling 1978; Munn 1979). From a scientific perspective, it has been widely observed that matrices and checklists do not provide explicit investigative protocols for predicting the biophysical impacts of development alternatives. This observation has generally been associated with the fact that matrices and checklists cannot depict spatial and temporal dimensions of the environment, or functional relationships among environmental entities (a problem first addressed by causal network diagrams). Instead, matrix and checklist approaches have generally been seen to favour the production of shallow and descriptive EAs, a phenomenon widely attributed to the lists of generic environmental components used to guide field surveys and data collection. Indeed, the compartmentalization of environmental entities into discrete and unrelated categories was seen to discourage interdisciplinary collaboration among specialists, further contributing to the production of fragmented and descriptive EAs. Overall, it was concluded that dimensionless impact scores are based on implicit expert opinion, rather than observable properties of the biophysical environment, and therefore do not represent valid or testable measures of environmental impact.

From both scientific and administrative perspectives, Andrews (1973) argued that comprehensive EA methods such as the Leopold matrix were the product of unrealistic expectations held by administrators for complete environmental information. In general, he argued that early regulatory approaches to implementing EA were often based on the assumption that more information alone leads to better decisions, a notion widely reinforced by the threat of judicial challenge. More specifically, Andrews (1973) argued that the Leopold matrix—being unable to generate explicit impact predictions using logical protocols—had failed to provide decision-makers with the critical insights needed to choose carefully among development alternatives. He concluded, therefore, that such early methods were unable to fulfill EA's intended role in fostering development decisions that protect important environmental values.

Early methods for conducting EAs are problematic not only from a scientific perspective, but from a political perspective as well. For example, Bisset (1978) has argued that by failing to decouple the quantification of impact significance (based on implicit expert opinion) from the estimation of impact magnitude (based on perceptible attributes of the environment), early matrix and checklist methods serve to impinge on the interpretation and evaluation of assessment results by other EA participants. In other words, by concealing the value-based judgements of those who prepare assessments in the form of numerical impact scores, the use of matrix and checklist methods ultimately serves to constrain wider stakeholder influence over development decisions.

4.3 First Generation Scientific Guidance

The first generation of scientific guidance materials for EA emerged in the late 1970s (e.g., Holling 1978; Ward 1978; Munn 1979). Such guidance collectively advocated an approach to EA based on collaborative problem-structuring, simulation modelling, quantitative measurement, and experimental manipulation. The most influential of these early frameworks—outlined by Holling (1978) and colleagues at the University of British Columbia—is composed of a set of concepts, protocols, and techniques collectively referred to as 'adaptive environmental assessment and management' (AEAM). To overcome reactive, fragmented, and descriptive approaches to EA, Holling's (1978) framework sought to coordinate ongoing communication and collaboration among scientists and EA practitioners beginning in the early stages of development design. To this end, the AEAM approach employs workshops and computer simulation modelling exercises to explicitly define problems surrounding proposed development alternatives. During early workshops, participants reach agreement on the most important components and relationships characterizing the ecosystems under study. Participants must also select perceptible attributes of key components—referred to as performance indicators—to serve as measures of environmental change. As workshops progress, implicit understandings of cause-effect relationships are gradually quantified and translated into dynamic computer simulation models. These models are to provide a shared conceptual and technical framework for cause-effect research, impact prediction, and ongoing effects monitoring programs.

The immediate objectives of the AEAM approach are twofold: (i) to formulate defensible impact predictions that are useful to decision-makers, and (ii) to explicitly acknowledge and reduce scientific uncertainty over time through continuous effects monitoring and model refinement. To effectively bridge functional gaps between development design, impact assessment, and regulatory approvals processes, AEAM workshops require close and ongoing collaboration between scientists, proponents, and regulators. In this way, EA is understood as an ongoing, collaborative, and focused approach to development design and environmental decision-making, rather than a reactive, fragmented, and descriptive approach to evaluating a single preferred development design.

4.4 Provisions for Scoping

In the late 1970s, attempting to overcome the proliferation of unfocused and overly descriptive EAs, the US Council on Environmental Quality (CEQ) outlined the first formal requirements for scoping—an early and open process for determining the specific issues to be addressed in an EA (CEQ 1980). Despite the provision of scoping requirements, and indeed the ongoing elaboration of scientific protocols for EA during the 1980s (e.g., Fritz et al. 1980; Duinker and Baskerville 1986; NRC 1986a; Walters 1986; Duinker 1989), scholars and practitioners continued to observe considerable weaknesses in the application of science in EA practice (e.g., Rosenberg et al. 1981; Clark et al. 1983; Bisset 1984; Culhane et al. 1987; Fairweather 1989). In general, it was observed that mainstream EA practice had continued to pursue comprehensive but superficial coverage of environmental entities, regardless of their relevance to development decision-making. It was also widely observed that impact predictions, if made at all, were generally comprised of vague statements about the likelihood of certain environmental conditions prevailing, with little or no accompanying discussion of uncertainty.

To explain the growing state of confusion and frustration surrounding the scientific adequacy of EA practice, Carpenter (1983) pointed to a major divergence of expectations and capabilities within the EA community. From both scientific and administrative perspectives, Carpenter (1983) observed that those responsible for reviewing EAs (administrators, legal professionals, public interest groups, and other intervenors) had developed unrealistic expectations of what could reasonably be accomplished by those responsible for preparing EAs (proponents and consultants). More generally, he observed that the wider EA community had thus far failed to acknowledge (and agree on coordinated measures to reduce) the profound uncertainty surrounding complex ecosystems. Instead, he observed that most EA administrators, legal professionals, and laypeople had come to expect complete and verified impact predictions from a fledgling environmental science operating within severe time and resource constraints.

Most importantly perhaps, Carpenter (1983) observed that the role of research scientists—i.e., those with the best understanding of ecological science and its practical limitations—was typically restricted to providing expert testimony during adversarial review hearings. The result, concluded Carpenter (1983), was a frustrating state of affairs in which research scientists were being asked to challenge or defend the quality of work prepared by proponents and their consultants, all within a judicial or quasi-judicial review framework that itself emphasized the legal and procedural requirements of EAs.

To reconcile conflicting expectations within the EA community, and share the burden of reducing uncertainty surrounding complex ecosystems, Carpenter (1983) called for: (i) the design and implementation of long-term ecosystem monitoring programs, undertaken collaboratively by governments, proponents, and research scientists, and (ii) the establishment of formal scientific advisory and review committees, intended to assist in the development and review of targeted research programs for particular EAs. In sum, Carpenter (1983) called for early and ongoing collaboration among all EA participants, including members of the scientific community.

4.5 Second Generation Scientific Guidance

In the 1980s, a second generation of scientific guidance materials—some North American (e.g., Beanlands and Duinker 1983; Duinker and Baskerville 1986; Duinker 1989) and some European (e.g., ERL 1981; ERL 1984; ERL 1985)—provided further insight into the implementation of science in EA. In a major Canadian report on the subject, Beanlands and Duinker (1983) outlined a series of six so-called requirements for conducting ecological studies in the context of formal EA:

i) Identify the VECs on which the analysis will focus
ii) Define a context for impact significance
iii) Establish boundaries for analysis
iv) Develop and implement a study strategy
v) Specify the nature of predictions
vi) Undertake monitoring

All six 'requirements' pertain to the planning and design stages of the EA process because, according to Beanlands and Duinker (1983), it is during these early stages that scientific improvements are most effectively realized. In summary, all EAs should be required to: (i) "identify at the beginning of the assessment an initial set of valued ecosystem components to provide a focus for subsequent activities"; (ii) "define a context within which the significance of changes in the valued ecosystem components can be determined"; (iii) "show clear temporal and spatial contexts for the study and analysis of expected changes in valued ecosystem components"; (iv) "develop an explicit strategy for investigating the interactions between a project and each valued ecosystem component, and to demonstrate how the strategy is to be used to coordinate the individual studies undertaken"; (v) "state impact predictions explicitly and accompany them with the basis upon which they were made"; and (vi) "demonstrate and detail a commitment to a well defined programme for monitoring project effects".

Crucial to focusing all subsequent stages of the research program is the identification of VECs through scoping. Indeed, the early identification of VECs provides an essential opportunity for all EA participants to build consensus around the most important environmental values at stake. In this way, practitioners are encouraged to design and implement targeted research programs aimed at providing useful and defensible insights into stakeholder-relevant issues.

According to Beanlands and Duinker (1983), adoption of the above requirements would not necessarily imply increasing costs or resources associated with EAs, but rather a reorganization of effort surrounding their design and conduct. To facilitate and encourage widespread adoption of the requirements, Beanlands and Duinker (1983) outlined four recommendations pertaining to organizational and administrative aspects of EA:

i) Adoption of the requirements by regulators, proponents, and consultants
ii) Establishment of scientific advisory committees
iii) Incorporation of monitoring into regulatory guidelines
iv) Increased involvement of the scientific research community

Of the four recommendations, (i) and (iii) call for simple changes to administrative procedures and requirements for EAs. More importantly, however, recommendations (ii) and (iv) advocate changes to the organizational arrangements for designing, conducting, and reviewing EAs. Beanlands and Duinker (1983) noted that incorporation of the scientific requirements into regulatory frameworks—while an important first step towards their implementation—would not alone suffice, as general guidelines would still require detailed scientific interpretation appropriate to individual assessments. Consequently, Beanlands and Duinker (1983) recommended that regulatory agencies administering EAs establish a small group of technical advisors—comprised of scientists from universities and government agencies—for each EA undertaken. The advisory group would be expected to work with proponents and consultants to develop a mutually-agreeable investigative strategy prior to the commencement of individual studies. The group of advisors would also be important participants in the final review of assessment results. In this way, EA preparers and reviewers would have an early opportunity to reach agreement on the most appropriate research design prior to the commitment of time and resources, thereby avoiding costly and frustrating reviews upon completion of the assessment. Moreover, by sharing some of the responsibilities for study design and review with independent advisory groups, the perceived impartiality and credibility of the EA process could be progressively fostered and maintained.

Since then, the Beanlands and Duinker (1983, 1984) proposals have been widely cited in both scholarly and practitioner literatures alike. While it is clear that the VEC concept has become an important device for focusing scientific investigations in EA practice, uptake of the other five requirements by the practitioner community appears to have been sparse. In the remainder of this paper, I build on the scientific foundations of the 1970s and 1980s by comprehensively reviewing and synthesizing insights from the relevant peer-reviewed formal literature.

CHAPTER 5 BEYOND TRADITIONAL SCIENCE

What kinds of science are appropriate in the context of EA? How can science create useful and defensible knowledge amidst such pervasive conflict and uncertainty? While the science of air, water, soil, and biota will inevitably make important contributions to most EAs, how can these disciplines work together to produce reliable knowledge in a complex and changing world? In this section I outline four approaches to scientific inquiry that have moved beyond the limitations and expectations of traditional 'bench science'. Here I describe how 'adaptive management', 'post-normal science', the 'transdisciplinary imagination', and 'citizen science' all provide mutually compatible frameworks for designing and implementing rigorous scientific investigations in the context of formal EA.

5.1 Adaptive Management

The concept of 'adaptive management' was originally outlined by Canadian fisheries scientists in the early 1970s (e.g., Walters and Hilborn 1976; Peterman 1977). In general, they observed that complex ecological systems—including the effects of human activities—were shrouded in considerable uncertainty, something that was not being considered in the government regulation of fisheries harvests. Moreover, they observed that most fisheries were being modelled and managed based on the assumption that fish stocks—in the absence of human intervention—would remain in a state of static equilibrium. Drawing on the work of ecological theorist Buzz Holling (e.g., Holling 1966; Holling 1973), Walters and Hilborn (1976) and Peterman (1977) argued that complex ecosystems are not static, but dynamic, and that they are typically characterized by multiple ecological processes, 'stability domains', and response thresholds. Overall, they concluded that simulation modelling is a highly useful tool for exploring the future consequences of alternative developments, but that some observation and measurement is ultimately needed in order to reduce key uncertainties and detect environmental changes. Through the process of 'adaptive management', such newfound knowledge would then be communicated to managers, regulators, and other stakeholders in order to inform corrective action (e.g., change harvest rates).

Carl Walters and Ray Hilborn later joined Buzz Holling (1978) and colleagues at the University of British Columbia and the International Institute for Applied Systems Analysis to outline a framework for adaptive environmental assessment and management. As previously mentioned, the so-called 'AEAM approach' would rely on a series of brief, interdisciplinary workshops and computer simulation modelling exercises to collectively define and evaluate a set of development alternatives based on the predicted responses of environmental variables (i.e., 'performance indicators'). The 'adaptive management' aspect of AEAM would then rely on the monitoring of environmental responses following development in order to reduce key uncertainties in predictive models and inform subsequent management actions. The long-term management of environmental effects and the ongoing reduction of scientific uncertainty through monitoring are therefore considered to be a single integrated endeavour, requiring close collaboration between researchers and practitioners. While Holling's (1978) book established a strong conceptual basis for AEAM, Walters' (1986) book would later expand on the mathematical basis for adaptive management.

Holling (1978) briefly indicated that "Adaptive management can take a more active form by using the project itself as an experimental probe". Walters and Hilborn (1978) would later elaborate on this statement by defining the terms 'passive' and 'active' adaptive management. According to Walters and Hilborn (1978), passive adaptive management uses "general-processes [sic] studies and previous experience with similar systems to construct the best possible prior model, then manage as if this model were correct while expecting some mistakes that can be used to improve the model as management proceeds". With respect to an active approach, Walters and Hilborn (1978) wrote that active adaptive management treats "all management actions as deliberate experiments that (if properly designed) will have a dual effect of simultaneously producing short-term yields and better information for long-term management." Broadly speaking, they concluded that passive adaptive management may eventually yield some useful knowledge, but that such an approach wastes time as it does not generate much insight into the behaviour of exploited ecosystems. According to Walters and Hilborn

(1978), “When we can break out of the passive mode and learn to treat the acquisition of functional information and indeed the whole management process as fundamentally experimental activities requiring active planning and judgment, then we may begin to talk about a *science* of ecological management”.

In the early 1980s, a major review and evaluation of AEAM (ESSA 1982) found that the approach had been applied in a variety of environmental management contexts since the late 1970s. The report also found, however, that the implementation of AEAM had been limited primarily to the early workshop and modelling stages. Simply put, the iterative ‘adaptive management’ principle of AEAM had yet to be fully realized. As a fundamental barrier to implementation, ESSA (1982) identified the tendency of institutions and even individuals within organizations to resist innovation and change (i.e., adaptation). Perhaps even more fundamentally, ESSA (1982) observed the tendency of individuals and organizations to resist the acknowledgement of uncertainty. Overall, the ESSA (1982) report concluded that the implementation of AEAM is as much a political endeavour as a scientific one, requiring the participation of: (i) key individuals within stakeholder organizations (i.e., ‘wise persons’) to champion and sustain long-term adaptive management programs, (ii) research scientists with relative financial and institutional independence from proponents to focus on reducing uncertainty, and (iii) public stakeholders to ensure transparency in the design, evaluation, and selection of development alternatives.

Despite scientific and political challenges surrounding early implementation efforts, a number of authors (e.g., Everitt 1983; Jones and Greig 1985; Mulvihill and Keith 1989) continued to praise the AEAM approach during the 1980s, highlighting its applicability to all stages of the general EA process (e.g., scoping, prediction, monitoring). From a scientific perspective, Jones and Greig (1985) wrote that “Reducing uncertainty is the primary objective of all science”. They then went on to write that:

> In a recently completed analysis of environmental impact assessment in Canada in an ecological context, Beanlands and Duinker (1983) strongly advocated a more scientific approach to assessment. Their position was that a major part of an assessment should be formulation of specific hypotheses of impact. Testing these

should form the basis of the assessment studies, and should continue after the project has begun. This is exactly what the AEAM approach advocates.

Despite their closing observation that "the AEAM technique has yet to be applied throughout a major EIA [environmental impact assessment] project", Jones and Greig (1985) concluded that adaptive management was still highly relevant in addressing many of the key objectives of the EA process. Consequently, a number of authors (e.g., Mulvihill and Keith 1989) began to focus more closely on the institutional and organizational arrangements needed to implement adaptive management in the context of formal EA.

In his influential book on pursuing sustainable development, Kai Lee (1993) revisited the science of adaptive management, explaining the approach using the metaphor of "a compass: a way to gauge directions when sailing beyond the maps". For Lee (1993), the science of adaptive management would provide a means of reducing critical uncertainties in the pursuit of sustainable development. From a political perspective, Lee (1993) also recognized that conflicts over stakeholder values and perspectives were an increasingly important barrier to achieving mutually agreeable (and sustainable) environmental decisions. Lee (1993) therefore proposed the marriage of adaptive management (Holling 1978; Walters 1986) and the consensus-based politics of principled negotiation. Referring to the politics of negotiation, Lee (1993) wrote: "Democracy, with its contentious stability, is a gyroscope: a way to maintain our bearing through turbulent seas". With respect to both adaptive management and principled negotiation, Lee (1993) concluded that "Compass and gyroscope do not assure safe passage through rough, uncharted waters, but the prudent voyager uses all instruments available, profiting from their individual virtues."

More recently, the literature describes the application of AEAM in a range of environmental management contexts, highlighting a number of noteworthy successes and failures. Examples include the adaptive management of forests (e.g., Haney and Power 1996; Stankey et al. 2003; Bormann et al. 2007), watersheds (e.g., Grayson et al. 1994; Gilmour et al. 1999; Allan and Curtis 2005), rivers (e.g., Walters et al. 2000; Meretsky et al. 2000; Susskind et al. 2010; Susskind et al. 2012), fish (e.g., Hilborn 1992; Volkman and McConnaha 1993; Smith et al. 1998; Walters 2007), wildlife (e.g., Williams and Johnson 1995; Williams et al. 1996; Nichols et al. 2007), wetlands (e.g., Weinstein et al. 1996; Walters 1997; Gunderson and Light 2006), and coral reefs (e.g., Hughes et al. 2000; McCook et al. 2010). Whereas some authors (e.g., Walters 1997) have continued to lament the scientific challenges of implementing an adaptive approach to environmental assessment and management, many others (e.g., McLain and Lee 1996; Gunderson 1999; Lee 1999; Shindler and Cheek 1999; Allan and Curtis 2005; Stringer et al. 2006; Allan et al. 2008; Walkerden 2005; Allen and Gunderson 2011; Greig et al. 2013) have continued to elaborate on the political and institutional challenges of doing so.

While there is ample evidence of AEAM's application within the sphere of renewable resource management, evidence of its implementation in regulatory project-EA appears to be sparse. Consequently, a number of authors (e.g., Carpenter 1997; Noble 2000; Thrower 2006; Canter and Atkinson 2010; Benson and Garmestani 2011) have continued to call for its implementation within the context of formal EA. According to Noble (2000), "Although adaptive management has yet to be fully integrated and tested in the context of EIA, […] principles and applications of adaptive management offer many new possibilities for strengthening the environmental assessment process".

Holling (1978) wrote that "Adaptive management is not really much more than common sense. But common sense is not always in common use". ESSA (1982), however, observed that "There are probably as many definitions of AEAM as there are persons who have been exposed to it". To clarify, ESSA (1982) outlined the three major components of AEAM: (i) concepts of adaptive management, (ii) methods of systems analysis, and (iii) procedures of modelling workshops. To this they added that "Confusion arises when different aspects of these components are emphasized in any one application". In my view, Holling's (1978) framework for AEAM represents a foundational set of principles, protocols, and techniques that *together* provide a strong basis for applying both empirical and deductive biophysical science in EA.

5.2 Post-Normal Science

The foundations for what would later become known as 'post-normal science' were laid by philosopher of science Jerry Ravetz in the early 1970s. In his book titled *Scientific Knowledge and its Social Problems*, Ravetz (1971) argued that modern human societies had thus far benefited tremendously from industrial and technological science, but that such forms of science had also begun to create newer and more challenging problems (e.g., environmental, human health issues). Such problems, he argued, were of a fundamentally different nature than the kinds of problems solved by traditional science. Not only were these new kinds of problems characterized by a high degree of scientific uncertainty, they were imbued with human values and emotions. Ravetz (1971) went on to describe what he saw as the collective search for, and emergence of, a new kind of science that would provide better answers to the seemingly insurmountable social and environmental issues of the day. According to Ravetz (1971), this new, innovative, and much needed mode of inquiry would be called 'critical science'. In explaining the merits of this new and emerging brand of science, Ravetz (1971) wrote that "The work of enquiry is largely futile unless it is followed up by exposure and campaigning; and hence critical science is inevitably and essentially political". Though he observed that the conventional standards of scientific acceptability would be inappropriate for judging the products of this newer kind of science, Ravetz (1971) concluded that "the establishment of criteria for adequacy for solved problems is possible, for the work will frequently be an extension and combination of established fields for new problems, and so critical science can escape the worst perils of immaturity."

In the 1980s, Jerry Ravetz teamed up with philosopher Silvio Funtowicz to elaborate on his new conception of science, this time in terms of 'total environmental assessment'. According to Funtowicz and Ravetz (1985), complex policy-related science could be differentiated into three broad categories based on two dimensions: system uncertainties and decision stakes. Ravetz (1986) wrote that "when both dimensions […] are low, we have what we may call applied science; straightforward research will produce a practical band of values of critical variables which the ordinary political processes can operate to produce a consensus". With respect to the second type of research, Ravetz (1986) wrote:

> When either dimension becomes moderately large, a new situation emerges; we call it technical consultancy. This is easiest to see in the case of system uncertainty; the consultant is employed precisely because his or her unspecifiable skills, and his or her professional integrity and judgment, are required for the provision of usable knowledge for the policy process.

With respect to 'high' levels of conflict and uncertainty, Ravetz (1986) wrote: "Passing to the more intractable case, where either dimension is very large, we have what we call a total environmental assessment. For here, nothing is certain, there are no boundaries or accepted methods for solving problems; the problem is total in extent, involving facts, interests, values, and even lifestyles".

Broadly speaking, Ravetz (1986) argued that the role of all policy-related science should be to contribute 'useful knowledge' that informs discussions and debates in the political domain. He also argued, however, that what is unknown is often more important than what is known. He proposed that in all kinds of policy-related research, it should also be the role of science to provide participants with 'useful ignorance'. In other words, beyond characterizing what is known, scientific contributions to decision-making should also characterize and communicate what is not known. In cases of 'total environmental assessment', however, Ravetz (1986) observed that a complete lack of knowledge (i.e., ignorance) precluded the use of traditional statistical tools for uncertainty analysis. According to Ravetz (1986),

> Where ignorance is really severe, as in total environmental assessment, then it is involved in the problem in ways that are both more intimate and more complex. For if ignorance is recognized to be severe, then no amount of sophisticated calculation with uncertainties in a decision algorithm can be adequate for a decision. Nonquantifiable, perhaps nonspecifiable, considerations of prudence must be included in any argument.

As before, Ravetz (1986) argued that this new kind of research—faced with tremendous scientific and ethical uncertainties—could not realistically be judged against the same criteria as other, more traditional forms of science. Ravetz (1986) concluded that if science and society are to persist in harmony with the biophysical environment, then a new kind of mutual learning would need to take place, one that would require (at minimum) agreed-upon criteria for evaluating the contributions of an emerging science operating in the face of considerable controversy and uncertainty.

Expanding on their framework for understanding and evaluating complex 'policy-related science', Funtowicz and Ravetz (1990) outlined the so-called 'NUSAP' scheme (Numerical, Unit, Spread, Assessment, Pedigree). Rather than relying on traditional statistical techniques, the NUSAP scheme would use numerical gradations to rank scientific contributions according to both system uncertainty and information quality. While the NUSAP scheme was designed to accommodate a range of quantitative and qualitative information, Funtowicz and Ravetz (1990) wrote that "Pedigree, finally, is the most qualitative and complex of all the categories of the NUSAP notational scheme. Its role is to represent uncertainties that operate at a deeper level than those of the other categories. It conveys an evaluative account of the production process of the quantitative information". In essence, the NUSAP scheme—a cornerstone of what would soon become known as 'post-normal-science'—was intended to clearly communicate how different types of uncertainty are related to the quality of scientific information. The end result, it was hoped, would be a shift from the traditional scientific focus on creating 'facts' to an emphasis on exploring the boundary between knowledge and ignorance.

Through a series of papers published in the early 1990s (e.g., Funtowicz and Ravetz 1991, 1992, 1993), the two authors would recast their views on 'total environmental assessment', eventually settling on the language of 'post-normal science'. Funtowicz and Ravetz (1993) again outlined three kinds of policy-related research based on two dimensions: system uncertainties and decision stakes. According to Funtowicz and Ravetz (1993), "One way of distinguishing among the different sorts of research is by their goals: applied science is 'mission-oriented'; professional consultancy is 'client-serving'; and post-normal science is 'issue-driven'". Here again, the authors argued that: (i) the uncertainties inherent in post-normal research are not amenable to the statistical techniques of error analysis used in traditional science, and (ii) the quality of contributions made by post-normal science cannot realistically be judged against the same standards as traditional science. According to Funtowicz and Ravetz (1993), post-normal science is “one where facts are uncertain, values in dispute, stakes high, and decisions urgent”. To foster quality contributions from this emerging form of science, Funtowicz and Ravetz (1993) argued for the participation of an “extended peer community”, one that would reach beyond the technical and professional specialists involved in traditional peer-review processes. Such widespread participation, they argued, would be needed to build consensus on such unique and complex issues. With respect to scientific quality and uncertainty in the ‘post-normal age’, Funtowicz and Ravetz (1993) concluded that “Now the task is to see what sorts of changes in the practice of science, and in its institutions, will be entailed by the recognition of uncertainty, complexity and quality within policy-relevant research”.

Petersen et al. (2011) have recently described the experience of implementing a post-normal approach to science in EA within the Dutch context. According to Petersen et al. (2011), a scandal in the late 1990s surrounding the management of uncertainty in EA practice resulted in the Dutch EA agency adopting guidelines for a post-normal approach to conducting and evaluating scientific contributions. Petersen et al. (2011) explain that the main allegations were that the agency’s EAs “leaned too much towards computer simulation at the expense of measurements” and “suggested too high an accuracy of the environmental figures published”. Petersen et al. (2011) also describe two sets of regulatory guidelines developed and issued by the Dutch EA agency in the early 2000s: one for managing scientific uncertainty and value plurality, and another for stakeholder participation. Petersen et al. (2011) then go on to describe and evaluate the application of the guidelines in a number of case studies, judging the ability of the guidelines to address three key elements of the ‘post-normal’ approach: (i) management of uncertainty, (ii) management of multiple perspectives, and (iii) extension of the peer community. Despite evidence of modest progress, the authors conclude that the institutional challenges of implementing practices of post-normal science through a government agency are considerable. In the words of Petersen et al. (2011), “perhaps other institutions are better suited to take the PNS [post-normal science] approach to its extremes".

In my view, the post-normal conception of science represents a strong set of principles for shifting EA's current focus on creating illusions of 'fact' to openly characterizing, communicating, and reducing key uncertainties. The application of post-normal science in EA therefore remains entirely consistent and compatible with the application of AEAM.

5.3 Transdisciplinary Imagination

In their seminal paper from the early policy science literature, Rittel and Webber (1973) observed the lay public's growing dissatisfaction with traditional academic sciences, particularly when faced with the more intractable social and environmental issues of the day. Indeed, the authors themselves questioned the ability of traditional scientific disciplines to resolve the unique kinds of problems faced by contemporary public policymakers. Rittel and Webber (1973) observed that unlike the well-bounded or 'tame problems' solved by more traditional and academic forms of science, the more ill-defined or intractable problems faced by planners and policymakers were of a fundamentally different nature. Rittel and Webber (1973) called these problems 'wicked problems'. According to Rittel and Webber (1973), wicked problems can be distinguished from tame problems in that they lack definitive formulations and solutions. In addition to being fundamentally unique, wicked problems are also laden with social values and perspectives, which are often in disagreement with one another. The set of potential solutions to a wicked problem therefore rests largely on one's perception of how that problem is structured.

Broadly speaking, Rittel and Webber (1973) argued that because resolving a wicked problem is essentially the same process as framing one, policy analysts should not decide on an 'optimal solution' too soon, as is often the case. The underlying paradox here is that the formulation of a wicked problem requires knowledge of potential solutions, which in turn depends on perceptions of problem structure. According to Rittel and Webber (1973), this is the main reason that traditional or academic science—

accustomed to neatly defined and agreed-upon problems and solutions—remains incompatible with contemporary wicked problems. The authors also went on to argue that wicked problems, unlike tame problems, are not amenable to scientific experimentation in the classical sense. Simply put, because wicked problems are unique, and because each attempt at a solution may cause irreversible social and environmental consequences, such problems cannot be experimented upon without penalty. Though the authors did not provide much guidance for tackling wicked problems, they did hint at a collaborative approach to structuring problems and deriving agreeable solutions. In the words of Rittel and Webber (1973),

> The systems-approach "of the first generation" is inadequate for dealing with wicked-problems. Approaches of the "second generation" should be based on a model of planning as an argumentative process in the course of which an image of the problem and of the solution emerges gradually among the participants, as a product of incessant judgment, subjected to critical argument.

Responding to growing criticism over the role of academic science in modern society, many universities and institutions of higher learning embarked upon programs of educational and organizational reform during the 1970s. Because traditional approaches to knowledge creation, organization, and dissemination were increasingly seen as fragmented and even irrelevant to solving society's most pressing problems, many universities sought to implement arrangements that would foster the conduct of more interdisciplinary research (e.g., Apostel et al. 1972). While many authors (e.g., Weidner 1973; Antonio 1978) were busy extolling the virtues of such interdisciplinary initiatives, others (e.g., Jantsch 1972; Piaget 1972) advocated a more ambitious transition to what they called 'transdisciplinarity'. According to Jantsch (1972), universities would be increasingly responsible for adopting a new primary purpose: aiding humanity to understand and enhance its collective capacity to persist and self-regulate in the face of rapid social and environmental change. In the words of Jantsch (1972), "the purpose of the university may be seen in the decisive role it plays in enhancing society's capability for continuous self-renewal". To this he added, "The new purpose implies that the university has to become a political institution in the broadest sense, interacting with government (at all jurisdictional levels) and industry in the planning and design of society's systems, and in particular in controlling the outcomes of the introduction of technology into these systems”.

Achieving this new purpose, argued Jantsch (1972), would require the design and implementation of new structures for organizing knowledge and conducting research within universities and within larger societies. Jantsch (1972) referred to these new organizational structures—or ‘integrated education/innovation systems’—as being ‘transdisciplinary’, and outlined a series of steps towards increasing cooperation and coordination between the disciplines. In ‘multidisciplinarity’, each discipline works within its own traditional boundaries. In ‘interdisciplinarity’, several disciplines work together in a related manner based on a set of common principles. In ‘transdisciplinarity’—the highest level of knowledge integration—research is organized around complex and socially-relevant problems that defy traditional disciplinary boundaries. Such transdisciplinary inquiry, argued Jantsch (1972) would be driven by society’s most pressing needs, rather than the curiosity of autonomous researchers.

The need to draw upon a variety of disciplines or domains of knowledge in environmental impact studies was originally recognized in the wording of early EA laws and policies (e.g., Cohen and Warren 1971; White 1972; Cramton and Berg 1973). Despite efforts to bring together multidisciplinary teams of experts and specialists to prepare the first EAs, early commentaries and reviews (e.g., Carpenter 1976; Schindler 1976; Ward 1978) criticized such attempts for producing little more than fragmented and descriptive reports. To encourage more meaningful communication and collaboration among the disciplines, Holling (1978) advocated a series of brief, intense workshops that would require a number of small teams—each comprised of subject matter experts, modellers, and support staff—to rely on the output of one another’s modelling efforts as input to their own modelling efforts. According to Holling (1978), through a process called ‘looking outward’, each subsystem modelling group (e.g., hydrology, vegetation, wildlife) is asked to consider possible links with other subsystems. In other words, each modelling team is asked what it would need from the other modelling teams in order to carry on its work. Such brief, interdisciplinary workshops, it was hoped, would help to overcome the sprawling and fragmented research which had been the hallmark of EA practice thus far.

More recently, the notions of 'wicked problems' and 'transdisciplinarity' have seen renewed currency in the context of environmental management and sustainable development (e.g., Scholz et al. 2000; Salwasser 2004; Thompson Klein 2004; Cundill et al. 2005; Pohl 2008; Chapin et al. 2008; Jentoft and Chuenpagdee 2009; Hughes et al. 2012; Mauser et al. 2013; Moeliono et al. 2014). In the recent book by Brown et al. (2010) titled *Tackling Wicked Problems through the Transdisciplinary Imagination*, the authors propose the integration and application of both frameworks in pursuing sustainable development. Explaining their choice of the term 'imagination', Brown et al. (2010) write: "imagination is associated with creativity, insight, vision and originality; and it is also related to memory, perception and invention. All of these are necessary in addressing the uncertainty associated with wicked problems in a world of continual change". They also go on to write that "imagination has been central to the work of anyone who is involved in change in the society in which they live [...]. It should come as little surprise that imagination plays an important role in decision-making on complex issues".

A number of authors (e.g., Bond et al. 2010; Greig and Duinker 2011) have recently argued for the implementation of transdisciplinary organizational arrangements and working practices in the context of formal EA. According to Bond et al. (2010), current sustainability debates go beyond the questions of multi- or inter- disciplinarity posed by earlier attempts at interpreting EA provisions. Simply put, growing demands for public participation and knowledge integration have pushed EA and related fields of inquiry into the realm of transdisciplinarity. Indeed, the conceptual and organizational framework proposed by Greig and Duinker (2011) for achieving stronger science in EA is couched in terms of Brown et al.'s (2010) 'transdisciplinary imagination'. In the words of Greig and Duinker (2011), "To the degree that we might see EIA problems as wicked problems (and surely they often are), we propose that scientific inquiry associated with EIA should be set into the context of the transdisciplinary imagination". Here I reiterate Greig and Duinker's (2011) proposal and observe the complementarity of such an approach with the other frameworks described so far.

5.4 CITIZEN SCIENCE

The term ‘civic science’ has long been used in the literature (e.g., Shen 1975) to mean the exchange of scientific information between researchers and the general public, particularly with respect to information that contributes to better public decision-making. Thus, civic science aims to improve the public’s understanding of important scientific knowledge as well as the ability of scientists to communicate that knowledge.

In the 1990s, two natural resource experts—one in Europe (Irwin 1995) and one in North America (Bonney 1996)—began referring to direct public participation in scientific research as ‘citizen science’. Each author’s use of the term, however, was slightly different. In general, Irwin (1995) observed society’s ongoing dissatisfaction and distrust with the professional conduct of policy-related scientific research. To improve relations between science and society for sustainable development, Irwin (1995) proposed that policy-related science processes be opened up for more direct public participation. So, while Bonney (1996) used the term ‘citizen science’ simply to mean the coordinated collection of environmental data (e.g., bird counts) by members of the lay public (e.g., birdwatchers), Irwin (1995) used it more ambiguously to mean both “science which assists the needs and concerns of citizens” and “a form of science developed and enacted by citizens themselves”.

In the context of formal EA, direct public involvement in data collection and interpretation is now commonly referred to as either ‘participatory monitoring’ or ‘community-based monitoring’ (e.g., Lawe et al. 2005; Hunsberger et al. 2005; Moyer et al. 2008). Likewise, public involvement in the predictive (i.e., deductive) stage of EA is now sometimes referred to as ‘participatory modelling’ (e.g., Videira et al. 2010; Bond et al. 2015). Broadly speaking, these authors argue that direct involvement of public stakeholders in the scientific stages of EA ensures greater overall transparency in the process, while also creating more opportunities for stakeholders to influence development decisions and to learn from one another. Similarly, a number of authors (e.g., Sinclair and Diduck 1995; Sinclair and Diduck 2001; Sinclair et al. 2008) have recently argued for the reconception of EA as a ‘social learning’ process, one that gives priority to stakeholder education about values, perspectives, environmental impacts, and the EA process itself.

This kind of social learning, it is argued, will inevitably require more informal opportunities for public participation during the design and conduct of EAs.

Overall, I find that inconsistent use of the terms ‘citizen science’ and ‘civic science’ has resulted in confusion surrounding their precise meaning, not to mention how they might be applied in the pursuit of sustainable development (see Clark and Illman 2001). I conclude, however, that the basic notions of science *for* the people (i.e., civic science) and science *by* the people (i.e., citizen science) are both generally applicable to the implementation of science in EA. Moreover, both concepts seem to fit reasonably well with the other frameworks outlined above.

CHAPTER 6 EMERGING CONCEPTS FOR SCIENCE IN EA

Since EA's inception in the early 1970s, several conceptual developments have taken place in the scientific literature. While some of these ideas (i.e., resilience, complexity, thresholds, landscape ecology) seem to be serving EA studies in the background, others (e.g., uncertainty, sustainability) have helped to define the fundamental purpose of EA as well as the role of science in achieving that purpose. Still other concepts (e.g., biodiversity, climate change) have helped to expand EA's traditional focus on stakeholder-relevant VECs that are normally identified on the basis of traditional ecosystem elements (e.g., particular species, specific ecosystems, air and water quality). In this section, I provide a summary of the most important pieces of literature that have helped to define these emerging concepts. I also provide a summary of any literature that has helped to establish relationships between these concepts and the conduct of scientific investigations in formal EA practice.

6.1 Resilience

The concept of resilience in ecological systems was first articulated by Canadian ecologist Buzz Holling. According to Holling (1973), the traditional 'equilibrium-centred view' of ecosystems—inherited from the physical sciences—could not adequately explain how ecosystems actually responded to natural and anthropogenic disturbances. Rather than simply returning to a single stable equilibrium, many ecosystems seemed to remain in an altered or degraded state, despite removal or cessation of the initial disturbance. Using two- and three-dimensional 'phase plane' diagrams, Holling (1973) showed how complex ecosystems may in fact be characterized by multiple 'domains of stability', the shape of each being determined by a particular set of parameter values. Whereas the equilibrium-centred view of ecosystems focused on the degree of constancy within a narrow band of stable (i.e., predictable) conditions, the resilience view of ecosystems would focus on the 'boundaries' (i.e., thresholds) separating multiple 'domains of attraction', each with its own unique stability characteristics. Broadly speaking, Holling's (1973) proposed resilience perspective would focus on the ability of ecosystems to absorb and even benefit from environmental variation and change.

Holling (1973) wrote that:

> Resilience determines the persistence of relationships within a system and is a measure of the ability of these systems to absorb changes of state variables, driving variables, and parameters and still persist […] Stability, on the other hand, is the ability of a system to return to an equilibrium state after a temporary disturbance. The more rapidly it returns, and with the least fluctuation, the more stable it is.

With respect to diversity, stability, and resilience, Holling (1973) concluded that—contrary to traditional thinking—ecosystems with more diversity (i.e., relationships) were in fact *less* stable than ecosystems with fewer relationships. He argued that it was this dynamic variability—the product of a long and stressful evolutionary history—that ultimately made such ecosystems so resilient. Overall, Holling (1973) concluded that equilibrium-centred approaches to resource management, which aim to reduce the natural variability of ecosystems (e.g., flood, fire control), actually serve to erode their resilience to change, slowly rendering them 'fragile' and vulnerable to sudden collapse. In other words, a slow quantitative change in parameter values, with no accompanying qualitative response, might suddenly give way to a dramatic 'flip' into another stable ecosystem configuration from which there is little hope of return. A resilience-based approach to ecosystem management, as proposed by Holling (1973), would instead focus on the need to maintain the spatial heterogeneity and temporal variability inherent in natural ecosystems, thereby preserving their ability to absorb future events and persist in the face of unexpected change.

As part of the emerging AEAM framework, Holling (1978) expanded on the notion of resilience to include the sociopolitical dimension. Holling (1978) proposed that human individuals, societies, and institutions—the counterparts of ecological systems and components—also exhibited properties of stability and resilience. According to Holling (1978), many large institutions, such as government agencies, businesses, and universities, were also being managed with the aim of reducing variability and controlling exposure to disturbance. Holling (1978) argued that all of this was done at the cost of maintaining responsiveness and adaptability in the face of unexpected change. In the words of Holling (1978),

> The ecological systems that have persisted have been those that were resilient enough to absorb the unexpected and learn from it. Our institutions, too, need a similar ability to cope. Institutions, like biological systems, learn to handle change by experiencing change. And as with other things learned, this ability will be forgotten if the experience is not occasionally reinforced. Insulation from small disasters leaves one ill-prepared and vulnerable to larger ones.

With growing concerns over the consequences of global climate change, Holling (1986) expanded on the notion of ecological resilience, using it to explain the relationship between local ecosystem complexity, global biogeochemical cycles, and sustainable development. First, he began by observing that vegetation succession, an important concept in understanding the dynamics of terrestrial ecosystems, had thus far only provided a partial picture of ecosystem dynamics. In other words, the prevailing theory of 'old field succession' had been conceived of simply as: (i) the rapid exploitation of abiotic resources by fast-growing pioneer species following a disturbance, and (ii) the gradual establishment and maturation of more competitive but slower growing climax species. To better understand the basic dynamics of ecosystems at an aggregate level, Holling (1986) proposed a figure eight-shaped cycle representing the sequential interaction of four aggregate ecosystem processes: (i) exploitation, (ii) conservation, (iii) creative destruction, and (iv) renewal. During the exploitation and conservation stages (the focus of traditional successional theory), ecosystems rapidly consolidate and organize matter and energy as the degree of connectedness between species and components increases. Eventually, things become too connected, homogeneous, and self-similar, and some kind of random disturbance event triggers the onset of 'creative destruction'. Here, stored capital is suddenly released, only to be exploited and reorganized all over again. Though Holling (1986) initially called this the ‘ecosystem cycle’, it would later become known as the ‘adaptive cycle’.

According to Holling (1986), the dynamics of creative destruction and renewal play an important role in shaping the stability and resilience properties of ecosystems, for it is here that materials and energy are made available for the creative forces of renewal. During these brief windows of opportunity, ecosystems are able to break old patterns and structures, reorganize, and adapt to evolving internal and external forces of change. It is also during these sudden releases, however, that ecosystems risk the irretrievable loss of material, forever altering the stability and resilience properties of such ecosystems. In the words of Holling (1986),

> Ecosystems have a natural rhythm of change the amplitude and frequency of which is [sic] determined by the development of internal processes and structures in a response to past external variabilities. These rhythms alternate periods of increasing organization and stasis with periods of reorganization and renewal. They determine the degree of productivity and resilience of ecosystems.

More recently, the notion of ecological resilience been elaborated upon in the context of other emerging concepts such as complexity (e.g., Gunderson and Holling 2001; Holling 2001; Puettman et al. 2009), biodiversity (e.g., Peterson et al. 1998; Gunderson 2000; Folke et al. 2004), and sustainability (e.g., Arrow et al. 1995; Folke et al. 2002). Making the connection between biodiversity and ecosystem resilience, Peterson et al. (1998) concluded that “ecological resilience is generated by diverse, but overlapping, function within a scale and by apparently redundant species that operate at different scales. The distribution of functional diversity within and across scales allows regeneration and renewal to occur following ecological disruption over a wide range of scales”. Likewise, relating the notion of ecosystem resilience to that of sustainable development, Arrow et al. (1995) concluded that “If human activities are to be sustainable, we need to ensure that the ecological systems on which our economies depend are resilient”. Simply put, sustainable development relies on the maintenance of ecological resilience, which in turn relies on the preservation of biodiversity.

At the same time, other authors (e.g., Gunderson 1999) have continued to elaborate on the relationship between resilience and adaptive management. According to Gunderson (1999), many of the observed failures of the AEAM approach can be attributed to a fundamental lack of resilience and flexibility in natural ecosystems, as well as in human societies. In the words of Gunderson (1999), "the successes and failures of AEAM are intertwined with system properties of flexibility and resilience. In a nutshell, if there is no resilience in the ecological system, nor flexibility among stakeholders in the coupled social system, then one simply cannot manage adaptively". In such cases, Gunderson (1999) concluded, efforts must be made to restore social and ecological resilience before adaptive management may successfully proceed.

The importance of adopting a resilience perspective in the context of formal EA has also been highlighted recently in the literature, both in terms of project-level (e.g., Benson and Garmenstani 2011; Bond et al. 2015) and strategic-level EAs (e.g., Slootweg and Jones 2011). From an administrative perspective, Benson and Garmestani (2011) argue that current EA laws and policies, if reconfigured, could provide an appropriate 'regulatory home' for adaptive management. The authors also observe, however, that existing environmental laws and regulations are typically rigid, having been established during a time when the 'equilibrium-centered view' of ecosystems prevailed. To successfully build resilience into ecological systems (and human institutions), Benson and Garmenstani (2011) argue for a reconfiguration and rewording of EA laws and policies to reflect the ideals of an iterative and adaptive approach environmental assessment and management.

From both scientific and political perspectives, Bond et al. (2015) highlight the importance of resilience thinking in EA, both in terms of ecological resilience as well as 'embedding evolutionary resilience' within the EA process itself. In the words of Bond et al. (2015), "uncertainty, ambiguity and ignorance […] can be better managed through embedding an evolutionary resilience approach, supported through participatory deliberation and adaptive management". To this they add, "Whilst designed for strategic IA in particular, there is no reason why such an approach cannot also be applied at the project level". Here I echo Bond et al.'s (2015) proposal for "embedding evolutionary resilience, participatory modelling and adaptive management" into formal EA practice. I also reiterate the urgent need to translate, test, and refine existing theory on the subject.

6.2 THRESHOLDS

The concept of response thresholds has had a long history in ecology, particularly with respect to the study of predator-prey interactions (e.g., Gause et al. 1936; Holling 1966; Larkin et al. 1964; Ware 1972). In general, the term was used by population/community ecologists to mean an abrupt, non-linear, and potentially irreversible change in the abundance or behaviour of animal populations within a particular predator-prey system. Similarly, concepts like assimilative capacity (e.g., Cairns 1977) and critical loads (e.g., Wong and Clark 1976) were used to conceptualize the ability of aquatic ecosystems to absorb or tolerate releases of anthropogenic pollutants. In theory, significant or irreversible environmental effects would be likely to occur if such limitations were exceeded.

Holling (1973) famously applied the notion of ecological thresholds to explain the general stability properties of complex ecosystems. According to Holling (1973), such complex ecosystems are typically characterized by multiple equilibria, each situated within its own unique 'stability domain'. Holling (1973) conceived of such 'stable ecosystem states' as being separated by theoretical boundaries called 'escape thresholds'. Should changes in ecosystem state variables or parameters exceed such thresholds, either due to planned human intervention or stochastic disturbance events, an ecosystem could potentially 'flip' or 'switch' into an alternate state of being from which it may be difficult or even impossible to return. Holling (1973) also referred more specifically to 'extinction thresholds', the density below which animal populations might cease to sustain themselves reproductively.

Holling (1986) later argued that the existence of ecological stability domains, as well as the boundaries (i.e., thresholds) separating them, could be attributed to a small number of ecological processes operating at different spatiotemporal scales, and hence at different speeds. Simply put, ecological stability domains (and the boundaries separating them) could be defined by the interaction of fast, intermediate, and slow variables nested within a spatial hierarchy. According to Holling (1986), it was the interaction of these three distinct speeds of variables that produced the cyclic variations typically observed in most ecosystems. More specifically, Holling (1986) wrote that “Jumps between the stability domains can be triggered by exogenous events, and the size of these domains is a measure of the sensitivity to such events”. To this he added: “The stability domains themselves expand, contract, and disappear in response to changes in slow variables. These changes are internally determined by processes that link variables and, quite independently of exogenous events, force the system to move between domains”. With respect to ecological thresholds, Holling (1986) concluded that “discontinuous change is an internal property of each system. For long periods change is gradual and discontinuous behavior is inhibited. Conditions are eventually reached, however, when a jump event becomes increasingly likely and ultimately inevitable”.

With respect to formal EA, the notion of ecological thresholds has been an important concept in determining the ‘significance’ of environmental impacts since the 1980s. Conover et al. (1985) in particular outlined a well-known approach to determining the significance of environmental impacts based on the application of ecologically-based thresholds. Such quantitative limits were intended to determine the assignment of qualitative significance rankings (e.g., ‘major’, ‘moderate’, ‘minor’, ‘negligible’) to predicted environmental impacts. Still other authors (e.g., Sassaman 1981; Haug et al. 1984) outlined somewhat more political approaches to impact significance determination based on the application of socially-derived ‘thresholds of concern’.

The notion of applying ecological thresholds to protect environmental values from the unwanted effects of human development has recently received increasing attention in the literature. While some authors (e.g., Toms and Lesperance 2003; Briske et al. 2006; Brenden et al. 2008; Sorensen et al. 2008; Andersen et al. 2009; Ficetola and Denoel 2009; Suding and Hobbs 2009; Samhouri et al. 2010) have outlined quantitative techniques for identifying and delineating ecological thresholds, others (e.g., Scheffer and Carpenter 2003; Bestelmeyer 2006; Groffman et al. 2006) have lamented the practical difficulties of doing so. Still others (e.g., Johnson 2013) have explored the fundamental differences and similarities between ecological, social, and regulatory thresholds. With respect to ecological thresholds and environmental decision-making, Johnson (2013) concluded that "Scientists […] will play the lead role in providing the technical information that is the ecological threshold, including a full accounting of uncertainty in such information and some perspective on the implications of conservation decisions that exceed those thresholds".

The use of thresholds for determining environmental impact significance has continued to receive considerable attention in peer-reviewed EA literature. While some authors (e.g., Duinker and Greig 2006; Duinker et al. 2013) have advocated the use of ecologically-based thresholds for determining the significance of environmental impacts, others (e.g., Canter and Canty 1993; Ehrlich and Ross 2015) have advocated the use of socially-derived thresholds of acceptability. Still others (e.g., Kjellerup 1999; Schmidt et al. 2008) have advocated the use of regulatory standards and thresholds for determining the significance of environmental impacts. Without a doubt, ecological response thresholds—if they can be identified—represent the maximum degree to which ecosystem components or processes can be altered and still be deemed sustainable. I argue, therefore, that such inherent biophysical limitations must provide the basis for all political and regulatory discussions surrounding environmental impact significance.

I conclude here with words of Duinker et al. (2013) who wrote:

> Perhaps the concept of thresholds represents the Achilles heel of CEA [cumulative effects assessment], and indeed perhaps all EIA, since thresholds are vital to determining impact significance in project-focussed EIA as well. There is a clear need for collaborative efforts of the scientific and policy communities to support implementation of helpful and influential CEA processes by establishing defensible thresholds for acceptable VEC conditions. Exemplary scientific progress in determining thresholds, no matter how tentative and uncertain, is sorely needed.

6.3 Complexity

The notion of complexity has had a long history in ecology. Here I cite early debates surrounding the so-called 'complexity-stability hypothesis' (e.g., MacArthur

1955; Elton 1958). Such arguments postulated that increases in trophic web complexity would lead to increases in community stability. This, it was argued, was due to greater redundancy of ecosystem functions. In other words, in the event that one trophic linkage was lost, many others would continue to provide that particular function, effectively buffering change and ensuring ecosystem stability. Other authors (e.g., May 1973; Holling 1973), however, challenged this traditional view of ecosystems, arguing that increases in ecosystem complexity would actually lead to greater ecosystem *in*stability. In Holling's (1973) view, complex ecosystems could be mathematically reduced to a set of key 'state variables' governed by changes in a related set of 'driving variables'. The form and function of the relationship connecting such variables could then be determined by estimating a set of 'parameters'. According to Holling (1973), it is the number, size, and shape of these functional 'stability domains' that determine the overall robustness (i.e., resilience) of ecosystems. In explaining the general characteristics of complex ecosystems, Holling (1973) pointed to multiple stable states, feedback loops, nonlinear thresholds, response lags, and density-dependent relationships. With respect to complexity, stability, and resilience, Holling (1973) concluded that:

> complex systems might fluctuate more than less complex ones. But if there is more than one domain of attraction, then the increased variability could simply move the system from one domain to another. Also, the more species there are, the more equilibria there may be and, although numbers may thereby fluctuate considerably, the overall persistence might be enhanced.

Broadly speaking, Holling (1973) concluded that it is precisely the unstable nature of complex ecosystems that allows them to persist and adapt in a changing world.

While Holling (1973) saw complex ecosystems as being fundamentally unstable and unpredictable, he also viewed their dynamic behaviour as being organized around stable 'domains of attraction'. May (1974), on the other hand, argued that the behaviour of complex ecosystems is often highly disorganized or even 'chaotic'. In such cases of chaos, May (1974) argued, ecosystems may be extremely sensitive to starting conditions, meaning that slight differences in input variables can result in vastly divergent or fundamentally unpredictable outcomes. In the words of May (1974), chaotic behaviour is "cycles of arbitrary period [sic] or aperiodic behaviour, depending on initial condition". This perspective on the behaviour of complex systems has since been dubbed 'chaos theory'.

Holling (1986) later expanded on his theory of complex ecosystems, this time emphasizing the hierarchical or cross-scale connections linking ecosystems operating at vastly different spatial and temporal scales. According to Holling (1986), local ecosystems could be best understood as being embedded within earth's global biogeochemical cycles. While the complexity of local ecosystems could still be captured mathematically in terms of state variables, driving variables, and parameters, driving variables in particular would have to be differentiated in terms of their 'internal' or 'external' orientation to the ecosystem in question. In this way, broad-scale phenomena such as global climate change could be understood as the 'emergent' effect of many small changes occurring at the local level. Conversely, the effect of global climate change on local ecosystems could also be seen as a kind of cross-scale feedback loop. In the words of Holling (1986),

> The increasing extent and intensity of modern industrial and agricultural activities have modified and accelerated many global atmospheric processes, thereby changing the external variability experienced by ecosystems. This imposes another set of adaptive pressures on ecosystems when they are already subject to local ones. As a consequence, locally generated surprises can be more frequently affected by global phenomena, and in turn can affect these global phenomena in a web of global ecological interdependencies.

In addition to addressing issues of scale, hierarchy, emergence, and feedback in complex ecosystems, Holling (1986) also sought to provide a synthesis of organized and disorganized notions of complexity. To this end, Holling (1986) introduced what he dubbed the 'ecosystem cycle' (later referred to as the 'adaptive cycle') to explain how complex ecosystems tend to alternate between periods of gradually increasing organization (i.e., renewal, exploitation, conservation) and periods of sudden disorganization (i.e., creative destruction). In the words of Holling (1986),

> The progression of events is such that these functions dominate at different times: from exploitation, 1, slowly to conservation, 2, rapidly to creative destruction, 3, rapidly to renewal 4, arid [sic] rapidly back to exploitation. Moreover, this is a process of slowly increasing organization or connectedness (1 to 2) accompanied by gradual accumulation of capital. Stability initially increases, but the system becomes so overconnected that rapid change is triggered (3 to 4). The stored capital is then released and the degree of resilience is determined by the balance between the processes of mobilization and of retention.

More recently, several authors (e.g., Gunderson and Holling 2001; Holling 2001) have elaborated on the relationships linking diversity, stability, productivity, and resilience in complex ecosystems. These authors have also extended their perspectives on resilience and complexity to include social and economic systems as well as ecological ones. Arguing that the term ‘hierarchy’ has become “so burdened by the rigid, top-down nature of its common meaning”, Holling (2001) “decided to look for another term that would capture the adaptive and evolutionary nature of adaptive cycles that are nested one within each other across space and time scales”. Ultimately, he would call such complex, adaptive, multi-scale systems ‘panarchies’. In the words of Holling (2001), “A panarchy is a representation of a hierarchy as a nested set of adaptive cycles. The functioning of those cycles and the communication between them determines the sustainability of a system”. Whereas the term ‘hierarchy’ implies top-down control of lower levels by higher levels, the term ‘panarchy’ implies that all levels or subsystems can influence one another. In the words of Holling (2001), “Each level is allowed to operate at its own pace, protected from above by slower, larger levels but invigorated from below by faster, smaller cycles of innovation. The whole panarchy is therefore both creative and conserving”.

From a somewhat more practical perspective, a number of authors (e.g., Kay et al. 1999; Pahl-Wostl 2007; Armitage et al. 2009; Puettmann et al. 2009) have outlined frameworks for explicitly managing forests and other renewable resources as complex adaptive systems. According to Puettmann et al. (2009), the overall aim of such an approach is to preserve ecosystem resilience and adaptive capacity, for it is the complexity and diversity of ecosystems that allows them to reorganize, innovate, and adapt in the face of sudden change. Simply put, the maintenance of ecosystem complexity is aimed at preserving the ability of ecosystems to evolve and persist into the distant future, a prerequisite for achieving sustainable development. For general understanding, Puettmann et al. (2009) outline six essential attributes of complex ecosystems: (i) nonlinear relationships, (ii) ill-defined boundaries, (iii) disequilibrium, (iv) self-organizing feedback loops, (v) emergent behaviours, and (vi) memory of past states. Puettmann et al. (2009) conclude with four key principles for managing forests and other renewable resources as complex adaptive systems: (i) consider a variety of ecosystem components, (ii) accept natural variability in space and time, (iii) actively maintain and develop heterogeneity in ecosystem structure, composition, and function, and (iv) predict and measure success at multiple scales.

With respect to the general applicability of 'panarchy theory' to pursuing sustainable development, Holling (2001) concluded that "The theory is sufficiently new that its practical application to regional questions or the analysis of specific problems has just begun". As both strategic- and project-level EA must inevitably deal with complex adaptive systems, I feel that complexity theory in general offers a powerful framework for understanding the dynamic behaviour of ecosystems without becoming too overwhelmed. Simply put, I feel that such theoretical frameworks provide a means of *simplifying complexity* while simultaneously capturing its essential elements.

6.4 LANDSCAPE ECOLOGY

The field of 'landscape ecology' as we know it today emerged during the 1980s, a time when ecological theorists were grappling with the confounding spatial dimension of complex ecosystems. The first landscape ecologists (e.g., Forman and Godron 1981; Forman and Godron 1986) sought to describe the different kinds of spatial structures and patterns commonly found in natural landscapes (e.g., 'patch', 'corridor', 'matrix').

Taking things a step further, other authors (e.g., Turner 1989) sought to understand the relationships linking such spatial patterns and structures to ecological processes operating at somewhat finer scales (e.g., wildlife population dynamics). Similarly, other authors (e.g., Shugart and West 1981; Urban et al. 1987) sought to understand the relationships linking scale, natural disturbance regimes, and spatial vegetation patterns within dynamic landscapes. Still other authors (e.g., Burgess and Sharpe 1981; Naveh 1982) sought to understand the effects of human development actions on particular landscape patterns and structures.

At the time of EA's genesis in the 1970s, landscape ecology was in its infancy (e.g., Troll 1971; Levin and Pain 1974; Levin 1976; Wiens 1976; Hansson 1977; Pickett and Thompson 1978). By the time of the Beanlands and Duinker (1983) study, the field was just beginning to flourish (e.g., Forman and Godron 1981; Burgess and Sharpe 1981). Although the establishment of formal EA processes and the development of landscape ecology occurred virtually concurrently, the emerging concepts and techniques of landscape ecology had not yet found major application in mainstream EA practice. Indeed, other domains of ecology were still prominent in EA, such as the analysis of food webs (e.g., Karr 1981; Hecky et al. 1984), nutrient cycles (e.g., Hecky et al. 1984), and wildlife habitat (e.g., Severinghaus 1981; Landres 1983; Hecky et al. 1984).

Later in the 1980s, a number of scientific guidance materials (e.g., Bedford and Preston 1988; Brinson 1988; Lee and Gosselink 1988; Weller 1988; Winter 1988) established the applicability of landscape ecology to assessing the cumulative effects of human development on the spatial structure, function, and diversity of terrestrial ecosystems, particularly forests and wetlands. In general, these authors argued that predicting and measuring the environmental impacts of human development at the 'local' or 'site' scale was simply not enough to infer the potential for cumulative effects of multiple developments occurring over much broader scales. In the words of Bedford and Preston (1988),

> a sound scientific basis for regulation will not come merely from acquiring more information on more variables. It will come from recognizing that a perceptual shift to larger temporal, spatial, and organizational scales is overdue. The shift in

> scale will dictate different--not necessarily more--variables to be measured in future wetland research and considered in wetland regulation.

Still other authors recognized the limitations of such a broad-scale spatial approach to assessing environmental impacts. In the words of Stakhiv (1988),

> The landscape ecology approach, however, is merely one part of the cumulative impact analysis puzzle. It resorts to the traditional habitat-based view of the environment. Despite the introduction of additional measures of spatial and temporal species needs, these measures are not especially helpful in gaining insights into the impacts of small-scale hydrologic and or water-quality perturbations.

In a related set of developments, the first spatially explicit landscape simulation models were created to predict the effects of human development on wetland and forest ecosystems (e.g., Kessel et al. 1984; Pearlstine et al. 1985; Costanza et al. 1988; Sklar et al. 1985). Since then, more sophisticated simulation models have been developed to predict the effects of timber harvesting, natural disturbance, and climate change on the structure, composition, diversity, and productivity of managed forest landscapes (e.g., Mladenoff and He 1999; Scheller and Mladenoff 2004; Scheller et al. 2007). Indeed, such models have found widespread application in the management of forests (e.g., Gustafson et al. 2000) and the conservation of threatened wildlife species (e.g., Akçakaya 2001; Larson et al. 2004).

Gontier et al. (2006) explored the possibility of using landscape simulation models in the context of formal EA. From a biodiversity perspective, the authors proposed the use of such models during the predictive and evaluative stages of EA processes. In the words of Gontier et al. (2006): "When impacts can be quantified and visualised, different landscape scenarios can be compared and evaluated from a biodiversity perspective. Further, vegetation succession can be modelled on a landscape level in order to consider long-term effects". With respect to empirical research, several recent studies have used biotelemetry techniques to quantify the effects of linear development structures (e.g., roads, pipelines, seismic lines) on habitat use by terrestrial mammals, particularly wolves and caribou (e.g., Dyer et al. 2002; Houle et al. 2009; Johnson et al. 2005; Polfus et al. 2011; Sorensen et al. 2008; Whittington et al. 2005).

In my view, these are welcome developments, as they demonstrate how landscape ecology can be used as a powerful set of lenses for understanding ecosystemic responses to development. I therefore expect to see the tools and methods of landscape ecology used prominently in ecological impact assessments in the near future. One potential reason for an imperfect fit, however, is the notion that EA is driven by stakeholder-relevant VECs that are normally identified on the basis of traditional elements of ecosystems (e.g., particular species, specific ecosystems, air and water quality). Indeed, it would seem that the concepts of landscape ecology (e.g., 'connectivity', 'patches', 'matrix') are not yet in the normal lexicon of most EA participants. I therefore hypothesize that landscape ecology may well be serving EA studies in the background but may not be found prominently in the documents directly associated with EAs.

6.5 Biodiversity

Modern notions of biodiversity can be traced back to earlier concepts like 'species diversity' and 'species richness' (e.g., Hurlbert 1971; Whittaker 1972; Peet 1974; Huston 1979). In the 1980s, many ecologists (e.g., Wilson and Peter 1988) began using the terms 'biodiversity' and 'biological diversity' to mean the variety of life at all levels of biological organization. While many authors (e.g., Myers 1988; Lugo 1988) continued to use the term biodiversity to mean simple species diversity, others (e.g., Burley 1988; Franklin 1988; Ray 1988) began using it more broadly to mean diversity of genetic material, species, functional groups, ecosystems, habitats, and landscape structures. In the words of Ray (1988),

> The goal of future efforts to address biodiversity must not be merely the compilation of lists of species. Though one must be sympathetic to intensive efforts to find out how much species diversity exists, there is no substitute for

learning how systems work, the implications of their characteristic diversity, and the role individual species play.

To quantify and protect biodiversity at multiple levels of organization, some authors (e.g., Noss 1987; Hunter et al. 1988) advocated a so-called ‘coarse-filter/fine-filter’ approach. Using this approach, a ‘coarse-filter’—representing a variety of important landscape structures, ecosystem types, community assemblages, and successional stages—is first applied to capture biodiversity in the broadest sense. Next, a ‘fine-filter’—comprised of habitat requirements for particular species—is applied to capture any rare or unevenly distributed species that may have ‘slipped through’ the coarse-filter. In the words of Hunter et al. (1988): “The conservation of biological diversity is too complex for monolithic approaches; and preservation of populations, species, communities, physical environments, ecosystems, and landscapes must all be considered, when deemed appropriate”.

In the early 1990s, the United Nations Earth Summit in Rio named the ‘conservation of biological diversity’ a major precondition for achieving long-term ecological resilience and sustainable development. According to Agenda 21 (UNCED 1992): “Urgent and decisive action is needed to conserve and maintain genes, species and ecosystems, with a view to the sustainable management and use of biological resources. Capacities for the assessment, study and systematic observation and evaluation of biodiversity need to be reinforced”. The report went on explicitly to identify formal project EA as a means of protecting Earth’s biodiversity from any unwanted effects of human development. According to the UNCED’s (1992) Agenda 21, governments, organizations, and societies should collectively:

> Introduce appropriate environmental impact assessment procedures for proposed projects likely to have significant impacts upon biological diversity, providing for suitable information to be made widely available and for public participation, where appropriate, and encourage the assessment of the impacts of relevant policies and programmes on biological diversity.

Since then, an abundance of scholarly articles and guidance materials have outlined frameworks for integrating biodiversity considerations into formal EA practice (e.g., Nelson and Serafin 1991; Nelson and Serafin 1992; CEQ 1993; CEAA 1996; Diaz et al. 2001; Slootweg and Kolhoff 2003; Geneletti 2003; Mandelik et al. 2005; Gontier et al. 2006; Wale and Yalew 2010; Slootweg et al. 2010; Brownlie et al. 2012). Recognizing a plurality of perspectives on the subject, Nelson and Serafin (1991) outlined a participatory approach to biodiversity assessment in which participants are invited to identify and discuss important components of biological diversity within the local or regional context. Such an approach to assessment, they argue, helps to ensure the production of scientific information that is useful and relevant to all EA participants. In the words of Nelson and Serafin (1991):

> Thinking about biodiversity in terms of environmental assessment provides a way of identifying and listing characteristics and parameters of biodiversity as information that is recognized by those living and working in the area. In this way, local knowledge, perceptions, values and activities related to past changes in environment and development can be integrated with biophysical surveys undertaken by specialists.

To this they added: "Such cooperative and participative approaches based on mutual learning and respect among those involved and affected are likely to prove essential to ensuring continuity in environmental management activities in the long run".

From a technical perspective, some authors (e.g., Gontier et al. 2006; Mörtberg et al. 2007) have recently explored the possibility of applying the tools and techniques of landscape ecology to predict the effects of human development on components of terrestrial biodiversity. According to Gontier et al. (2006):

> the creation and alteration of landscape scenarios in GIS makes it possible to predict and assess the impacts of planned developments and landscape changes on biodiversity components [...] This means that impacts of, for instance, habitat

fragmentation can be quantified and visualised, and alternative planning scenarios can be compared and evaluated from a biodiversity perspective.

Despite calls for the application of biodiversity concepts in formal EA, reviews of practice (e.g., Buckley 1995; Mandelik et al. 2005; Wegner et al. 2005; Gontier et al. 2006; Söderman 2006; Khera and Kumar 2010) have generally found the inclusion of such elements to be weak. As with the principles of landscape ecology, I suspect this may reflect an imperfect fit between assessments based on readily identifiable elements of ecosystems (e.g., species, air and water quality) and assessments based on abstract components of biodiversity (e.g., ecosystem types, habitat types, patch connectivity). Despite such differences, I expect to see the various components of biodiversity increasingly considered alongside more traditional ecosystem components in mainstream EA practice. In my view, the inclusion of biodiversity considerations in EA will help to protect the overall resilience of ecosystems and, ultimately, the sustainability of VECs.

6.6 Sustainability

The notion of 'sustainable development' was popularized in the late 1980s by the United Nations World Commission on Environment and Development (WCED 1987), which defined it as "development that meets the needs of the present without compromising the ability of further generations to meet their own needs". The WCED (1987) report went on to describe sustainable development as being: (i) contained within ecological limits, and (ii) socially agreeable and equitable. Simply put, sustainable development is that which conserves valuable environmental resources for continued use by present and future generations. The WCED (1987) also went on to explicitly identify formal EA as a means of pursuing sustainable development, arguing for the expansion of EA processes to consider the environmental effects of policies and programs as well as individual projects. According to the WCED (1987) report, "This broader environmental assessment should be applied not only to products and projects, but also to policies and programmes, especially major macroeconomic, finance, and sectoral policies that induce significant impacts on the environment".

Shortly after the WCED's (1987) declaration on sustainability, an abundance of papers was published on the role of formal EA in achieving sustainable development (e.g., Rees 1988; Gardner 1989; Jacobs and Sadler 1990). Drawing on the Beanlands and Duinker (1983) report, Gardner (1990) wrote: "Even traditional EIA, especially when strengthened and focused by the "ecological framework", plays an essential role in fostering awareness of the environmental consequences of development activities and of tradeoffs involved". To this she added: "A more flexible and widely applied approach to EIA could extend these benefits to other phases and levels of decision-making".

Since then, the notion of EA as a tool for pursuing sustainable development has been widely endorsed by scholars and practitioners alike (e.g., Sadler 1996; Lawrence 1997; Sinclair et al. 2008). At the same time, other authors (e.g., Pope et al. 2004; Gibson et al. 2005; Gibson 2006; Weaver and Rotmans 2006) have argued for a more 'integrated' approach to evaluating the social and environmental effects of proposed developments. They collectively refer to this new approach as 'sustainability assessment'. Gibson (2006) in particular has proposed a set of 'core generic criteria' and 'trade-off rules'—encompassing multiple dimensions of sustainability (e.g., social, environmental, economic)—to serve as a framework for assessment and decision-making. According to Gibson (2006), the overall aim of the approach is to make 'net' or 'positive contributions to sustainability' based on explicit trade-offs made among decision criteria. In the words of Gibson (2006): "Sustainability assessment is committed to positive overall contributions to a more desirable and durable future by identifying best options (not just acceptable undertakings) and multiple reinforcing gains (not mere avoidance of problems and mitigation of adverse effects)".

Still, other authors (e.g., Scrase and Sheate 2002; Kidd and Fischer 2005; Morrison-Saunders and Fischer 2006) have criticized such 'integrated' approaches to assessment for encouraging the comparison of environmental and socioeconomic considerations, hence legitimizing environmental trade-offs for economic gain, and the social and environmental benefits thought to flow from such gain. In the words of Morrison-Saunders and Fischer (2006):

> recent trends towards more integration, particularly in the context of sustainability assessment (SA) mean that social and economic aspects are now frequently considered on a par with the environment in impact assessment processes. There are indications that this development will ultimately favour trade-offs towards socio-economic benefits, causing adverse environmental impacts.

Overall, Morrison-Saunders and Fischer (2006) concluded:

> That current developments in SEA [strategic environmental assessment] and SA should downplay environmental issues is somewhat ironic and certainly a cause for concern since the driving force behind the development of EIA in the 1970s was to ensure that environmental factors were adequately considered prior to decisions on development proposals being taken. We argue that this need has not changed, and given the extent of national and global environmental degradation, if anything, is greater than it has ever been before.

Here I agree with the conclusion put forward by Morrison-Saunders and Fischer (2006), and rely on the definition of EA's central task offered by Duinker and Greig (2006): to contribute to sustainable development "by safeguarding VEC sustainability in the face of development that might compromise that sustainability". I further rely on Duinker et al. (2013) who recently concluded that: "the whole point of EIA, and indeed the reason CEA was embedded into it, is to make human activities more sustainable. That aim demands protection of important ecological (and social) values, represented in EIA by the concept of the VEC".

6.7 Climate Change

The notion that human activities may be driving changes in Earth's global weather patterns began to gain traction in the late 1980s (e.g., Ramanathan et al. 1985; Ramanathan 1988; Mitchell 1989). Indeed, some of the first scientific research exploring the potential ecological consequences of global climate change was conducted at this time (e.g., Clark 1988; Pastor and Post 1988; Smith and Tirpak 1989). However, it was not until the early 1990s—at the United Nations Earth Summit in Rio—that governments around the world began to take serious action on global climate change. Here, foundations were laid for an international agreement—the United Nations Framework Convention on Climate Change—aimed at the "stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" (UNFCCC 1992).

As part of the Rio climate convention, the UNFCCC (1992) identified two general kinds of implementable measures for dealing with global climate change and its effects: 'mitigation' and 'adaptation'. Simply put, mitigation measures would be aimed at minimizing the impact of human activities on global weather patterns by reducing greenhouse gas emissions. Adaptation measures, on the other hand, would be aimed at reducing social and ecological vulnerabilities to a changing climate through careful foresight and planning. In the words of the UNFCCC (1992), all state parties shall:

> Take climate change considerations into account, to the extent feasible, in their relevant social, economic and environmental policies and actions, and employ appropriate methods, for example impact assessments, formulated and determined nationally, with a view to minimizing adverse effects on the economy, on public health and on the quality of the environment, of projects or measures undertaken by them to mitigate or adapt to climate change.

Later, the original UNFCCC (1992) agreement would be expanded by the Kyoto Protocol (UNFCCC 1998), which would explicitly commit state parties to reducing their greenhouse gas emissions in ways that reflect national differences (e.g., wealth, current emissions).

Though the EA community was quick to endorse the notion of EA as tool for addressing global climate change in the 1990s (e.g., Robinson 1992; Sadler 1996; CEQ 1997), practical guidance on the subject has only recently begun to emerge (e.g., CCCEAC 2003; CEQ 2010; Argawala et al. 2010; Byer et al. 2012; Murphy and Gillam 2013). At the same time, reviews of practice (e.g., Sok et al. 2011; Slotterback 2011;

Wende et al. 2012; Watkins and Durning 2012; Chang and Wu 2013; Kamau and Mwaura 2013; Ohsawa and Duinker 2014; Jiricka et al. 2016; Enríquez-de-Salamanca et al. 2016) have highlighted a variety of challenges surrounding the incorporation of climate change considerations into EA practice, particularly at the project level.

In general, most scholarly reviews and guidance materials now acknowledge that it is impossible to predict how much global temperature would increase due to greenhouse gas emissions from a single project. For example, Canada's federal guidelines for incorporating climate change into EA (CCCEAC 2003) state that "unlike most project-related environmental effects, the contribution of an individual project to climate change cannot be measured". In other words, though it may be relatively simple to quantify the greenhouse gas emissions associated with a particular development using techniques like life cycle analysis (e.g., Odeh and Cockerill 2008) or carbon budget modelling (e.g., Kurz et al. 2009), the task of evaluating the 'significance' of such emissions continues to present a considerable challenge to practitioners. Indeed, many scholars (e.g., Christopher 2008; Slotterback 2011; Wende et al. 2012; Ohsawa and Duinker 2014) now agree that the gap between global or national emissions reduction targets and the emissions of local projects makes it very difficult to attribute any meaningful level of significance to such emissions. What is needed, they argue, is a kind of 'tiered' EA regime, whereby regional- or strategic-level EAs provide a sort of 'backdrop' for project-level EAs. In this way, the significance of project-level emissions may be more meaningfully evaluated in terms of local, regional, national, and global emissions reductions targets, and then mitigated accordingly (i.e., best available technology, compensatory measures).

With respect to current EA practice, a number of scholarly reviews (e.g., Kamau and Mwaura 2013; Ohsawa and Duinker 2014; Jiricka et al. 2016) have observed an overwhelming focus on climate change mitigation. Kamau and Mwaura (2013), for example, write that "EIA practice in Kenya just like in other countries of the world is yet to effectively integrate the climate change adaptation". Likewise, Jiricka et al. (2016) wrote that "To date, greater attention has been paid to climate change mitigation (reduction of greenhouse gases) than to the adaptation to climate change effects". Indeed, many authors (e.g., Slotterback 2011; Jiricka et al. 2016) have pointed out that a major challenge surrounding the incorporation of climate change into environmental impact predictions has been a high degree of uncertainty surrounding the precise local and regional consequences of a changing global climate.

To improve explorations of the future in cases of considerable uncertainty (e.g., climate change), Duinker and Greig (2007) proposed greater use of scenarios and scenario analysis in formal EA practice. In the words of Duinker and Greig (2007): "To understand deeply whether developments can be sustainable, we need to assess them against scenarios that provide sharp contrast in alternative futures. Each of the scenarios must be rooted in the present, plausible (not impossible), and internally consistent". With respect to global climate change, Duinker and Greig (2007) concluded: "We need to include in the set of an EIA's scenarios a comprehensive range of potential future developments, and all the key driving forces, such as climate change and human demographics, that can measurably affect the VECs".

In my view, project-level EA has the potential to become an important tool for addressing issues of global climate change, both in terms of mitigation and adaptation. I therefore expect to see climate change considerations increasingly incorporated into EAs conducted at the project level in the near future. I also anticipate, however, that if formal project EA is to contribute effectively to climate change mitigation efforts, it will need to be nested within broader EA regimes operating at regional or strategic levels. In this way, the 'significance' or sustainability of project-level emissions may be more meaningfully judged in the context of regional-scale emissions inventories and targets for reduction. At the same time, I note that project-level EA seems to be well suited to supporting climate change adaptation, since EA processes were originally intended to identify, evaluate, and select development alternatives based on a variety of environmental considerations. Simply put, it appears that project-level EA is already designed to incorporate the effects of climate change into impact prediction, effects monitoring, and adaptive management activities. Moreover, I point out that failing to consider the effects of climate change in project EAs may seriously invalidate environmental impact predictions. Therefore, despite the considerable degree of uncertainty surrounding the local environmental consequences of global climate change, I urge all practitioners of EA to acknowledge, characterize, and explore the uncertainties associated with climate change using innovative techniques like scenario building and scenario analysis.

6.8 Ecosystem Services

The notion of ecosystem services was popularized in the early 2000s by the United Nations Millennium Ecosystem Assessment, which defined them as "the benefits people obtain from ecosystems" (MEA 2003). The report went on to differentiate four broad categories of ecosystem services, namely: "provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services, such as nutrient cycling, that maintain the conditions for life on Earth". Overall, the notion of ecosystem services was intended to provide a conceptual framework for understanding critical relationships linking ecosystems, human well-being, and environmental decision-making. According to the report, "This enables a decision process to determine which service or set of services is valued most highly and how to develop approaches to maintain services by managing the system sustainably" (MEA 2003).

Since then, several authors (e.g., Geneletti 2013; Helming et al. 2013; Honrado et al. 2013; Kumar et al. 2013; Partidario and Gomes 2013; Rega and Spaziante 2013) have explored the possibility of using the ecosystem services concept to frame both strategic and project-level EA studies. While some authors (e.g., Karjalainen et al. 2013; Rosa and Sánchez 2016) have praised the approach for its emphasis on linking ecosystemic properties to human benefits and values, others (e.g., Baker et al. 2013) have argued that "the use of ecosystem services language may not resonate well with all stakeholders". However, most authors now agree that there has been limited experience applying the concept of ecosystem services in formal EA practice.

In addition to highlighting differences of opinion on the subject, Baker et al. (2013) delineated two broad approaches to incorporating ecosystem services in EA: comprehensive and philosophical. According to Baker et al. (2013), "The former is marked by the more quantitative approach to ecosystem services—this may include a systematic identification of ecosystem service supply and demand across an area and may extend to the valuation of ecosystem services", whereas "The ecosystem service philosophy is more about the use of ecosystem services as a heuristic or as a framing for the environment". With respect to the philosophical approach, Baker et al. (2013) concluded that "it is a less significant departure from existing practice and relies on a changing of language and elements of the approach".

Here we argue that EA's central task—protecting the sustainability of VECs—is entirely compatible with an ecosystem services perspective, warranted the chosen ecosystem services are of interest to some or all EA participants.

6.9 UNCERTAINTY

Holling (1978) argued that the purpose of environmental science, and indeed all science, is the reduction of uncertainty. Rather than trying to ignore or even eliminate scientific uncertainty—as is often the case in mainstream EA—Holling (1978) argued that it should be the role of researchers and practitioners of EA to explicitly identify, characterize, and reduce key uncertainties using a variety of quantitative and qualitative techniques. In the words of Holling (1978): "The future is uncertain. Few would disagree with this in principle […] environmental assessments are not, and cannot be, predictions in any real sense". To this Holling (1978) added:

> The people making environmental assessments often are the first to admit that their conclusions are not certain. But if they attribute their doubt to a lack of time, money, and manpower, then they have missed the point. Attempting to close the gap on imperfect predictions detracts from a proper focus on the consequences of the inherent uncertainties that will always remain. If prophecy is impossible, then go for understanding.

To provide a structured way of thinking about uncertainty in natural systems, Holling (1978) outlined three general types of uncertainty. The first class "involves those events that can be predefined, that have known direct effects, and that have known probabilities of occurrence"; the second class "involves those events that are imaginable and at least partially describable, but for which neither the outcome nor the probability of occurrence are known"; and the third class "contains all those events for which we have no experience (or have forgotten) and events involving unknown processes of unknown functional form". Walters (1986) would later outline a similar typology of uncertainty comprised of: (i) "background variation or noise", (ii) "statistical or 'parametric' uncertainty about the forms and parameter values of various functional responses", and (iii) "basic structural uncertainty about even what variables to consider". In the words of Walters (1986): "It is important to note that the distinction between noise, parameter error, and structural uncertainty does not stand up under close logical inspection. It is just a simple way of modeling a continuous spectrum of things that can go wrong". With respect to structural uncertainty, Walters (1986) concluded that: "I doubt that there can, in principle, be any consensus about how to plan for the inevitable structural uncertainties that haunt us, any more than we can expect all human beings to agree on matters of risk taking in general".

As a general approach to identifying and addressing key uncertainties during the predictive stage of EA processes, both Holling (1978) and Walters (1986) advocated the use of: (i) model invalidation, (ii) sensitivity analysis, and (iii) alternative models or predictions. In general, Holling (1978) argued that any attempt to compare a model's output to similar situations in the real world was an attempt to challenge or 'invalidate' the underlying logic of that model. Such an approach, Holling (1978) argued, would be an important way to first build confidence in a model's credibility.

With respect to parametric uncertainty, Holling (1978) wrote that:

> Frequently a particular relationship is not well understood or supported by data […] Although more data might help to resolve this problem, decisions and predictions must be made in the meantime. Rather than ignore the problem, or use a simpler model, it seems best to develop the model, and test it with several alternative assumptions. Sensitivity analysis is the term used for the process of testing the effects of different assumptions and parameter values on model predictions.

With respect to structural uncertainty, Holling (1978) wrote that: “A model could make all the testable predictions referred to above and still be the wrong representation of reality. The chance always exists that other models will meet these historical tests equally well but give very different predictions of future impacts or management success”. To this he added: “The greatest hope of any search for alternative models is always to find one that passes a greater number of significant invalidation tests than the original. Failures are almost as useful as successes”.

Ultimately, both Holling (1978) and Walters (1986) concluded that critical uncertainties would almost always remain surrounding the behaviour of complex ecosystems. The only reasonable way to reduce and manage such uncertainties, they argued, would be through a combination of active experimentation and passive monitoring of environmental effects following development implementation. In the words of Holling (1978): “Such an extension of activity requires the addition of a monitoring capability. At the very least, monitoring provides an opportunity to attempt an invalidation of the analysis that has already been done. Prediction may not be possible, but some postdiction is”.

Whereas Holling (1978) and Walters (1986) focused primarily on ‘measurable’ aspects of uncertainty (i.e., natural variability, parametric uncertainty), Ravetz (1986) chose to focus more on the intractable or ‘unmeasurable’ aspects of uncertainty (i.e., ignorance). Referring to what he called ‘total environmental assessment’, Ravetz (1986) wrote that: “For here, nothing is certain, there are no boundaries or accepted methods for solving problems; the problem is total in extent, involving facts, interests, values, and even lifestyles”. Ravetz (1986) further explored the relationship between full disclosure of scientific uncertainty and the resulting quality and usefulness of scientific contributions. In general, he concluded that if scientific contributions are to be useful to all participants of policy-science processes (like EA), then researchers and analysts must carefully characterize and communicate uncertainty in a manner that is transparent and comprehensible to all parties involved. Though Ravetz (1986) did not offer much advice on how ignorance and uncertainty should be handled in the context of EA, he did indicate that traditional statistical analyses would generally not suffice.

Later, Funtowicz and Ravetz (1987, 1990) would elaborate on the relationship between scientific quality and uncertainty, outlining a framework called the 'NUSAP scheme' for quantifying uncertainties in policy-related science. According to Funtowicz and Ravetz (1990),

> The NUSAP scheme was designed as a robust system of notations for expressing and communicating uncertainties in quantitative information […] NUSAP is an acronym for the categories Numeral, Unit, Spread, Assessment and Pedigree. Going from left to right, we proceed from more quantitative to more qualitative aspect of the information.

Elaborating on the five categories of NUSAP, Funtowicz and Ravetz (1990) wrote that: (i) "Numeral entry may be a number, or a set of elements and relations expressing magnitude"; (ii) "Unit represents the base of the underlying operations expressed in the numeral category"; (iii) "Spread category normally conveys an indication on the inexactness of the information in the numeral and unit places"; (iv) "Assessment expresses […] a judgement of the (un)reliability associated with the quantitative information conveyed in the previous categories"; and (v) "Pedigree […] conveys an evaluative account of the production process of the quantitative information".

More recently, the issue of scientific uncertainty (and how best to deal with it) has become a popular topic of discussion in the peer-reviewed EA literature. Generally speaking, reviews of theory (e.g., Rotmans et al. 2001; Walker et al. 2003; Refsgaard et al. 2007; van der Sluijs 2007; Warmink et al. 2010; Bond et al. 2015; Gregr and Chan 2015; Leung et al. 2015; Cardenas and Halman 2016) and practice (e.g., Tennøy et al. 2006; Duncan 2008; Wardekker et al. 2008; Petersen et al. 2011; Larsen et al. 2013; Leung et al. 2016; Lees et al. 2016) have highlighted major shortcomings in the characterization, disclosure, and ongoing reduction of scientific uncertainty in the context of formal EA.

As previously mentioned, Duinker and Greig (2007) have proposed greater use of scenarios and scenario analysis in EA to accommodate critical uncertainties in ecosystem drivers (e.g., climate change, market demand, future developments). In the words of Duinker and Greig (2007):

> Scenario-based approaches to forecasting environmental impacts offer a way to grapple with uncertainties inherent in predictive exercises that reach into the long-term future. If it is possible to launch serious challenges to relationships inside predictive models or to important contextual phenomena outside the model boundaries (i.e., challenges that would make us highly skeptical of the original forecasts), then scenario analysis is called for.

In my view, there are currently ample tools and techniques available to assist in the characterization, expression, and ongoing reduction of scientific uncertainty in the context of formal EA. In addition to well-known techniques like sensitivity analysis and scenario analysis, notational schemes like NUSAP may also be used to quantify and disclose the critical uncertainties associated with particular scientific contributions to EA. Still, I conclude that lingering uncertainties will inevitably always remain, necessitating an ongoing adaptive approach to environmental effects monitoring, active experimentation, and predictive modelling.

CHAPTER 7 SCIENCE IN THE EA PROCESS

7.1 Scoping

7.1.1 Overview

In EA processes, scoping refers to an early set of activities aimed at defining a stakeholder-relevant focus for assessments. Through scoping, professionals and stakeholders identify concerns surrounding proposed developments and agree on the most important environmental values at stake. As with the formal review stage of EA processes, scoping implies the convergence of political, scientific, and administrative expectations and perspectives. From an administrative perspective, scoping produces a list of proposal-specific guidelines that direct the proponent in preparing an environmental impact statement (EIS) or similar EA document. From a political perspective, scoping provides an early opportunity for all interested parties to influence the general design of assessments prior to their implementation and review. From a scientific perspective, scoping aims to focus assessments on a well-defined set of questions for which targeted research programs can be designed and implemented. Overall, the scoping process is aimed at fostering early agreement among professionals and stakeholders alike as to how an EA will proceed and on what basis scientific contributions to that EA will be evaluated during review.

Originally, formal EA processes did not require an initial scoping exercise to focus assessments. The tendency of these early processes was to produce voluminous and excessively descriptive documentation, a phenomenon which was repeatedly observed in the literature of the 1970s (e.g., Andrews 1973; Carpenter 1976; Schindler 1976; Holling 1978; Ward 1978; Munn 1979). Indeed, early frameworks for conducting EAs (e.g., Leopold et al. 1971; Dee et al. 1973) advocated comprehensive coverage of environmental entities, regardless of their relevance to development decisions. Subsequent scientific guidance materials (e.g., Holling 1978; Ward 1978; Munn 1979) and regulatory provisions (e.g., CEQ 1978) called for more focused EAs predicated on early and collaborative problem-structuring exercises. While Holling (1978) referred to such early and collaborative problem structuring as 'bounding', official EA guidelines (e.g., CEQ 1980) would later to refer to it as 'scoping'. In general, such guidance sought to shift the initial point of contact between stakeholders and professionals away from adversarial reviews to focus on more creative aspects of assessment planning and design.

In the 1980s, scientific guidance materials for EA (e.g., Beanlands and Duinker 1983; Beanlands 1988) thoughtfully elaborated on the principles of scoping outlined by administrators in the late 1970s. Most notably, Beanlands and Duinker (1983) coined the term 'VEC' to encourage those responsible for preparing EAs to focus their investigations on environmental entities of interest to public and professional stakeholders. Indeed, Beanlands and Duinker (1983) recognized that the immediate goal of EA—to protect environmental values by influencing development decisions—would be more likely to succeed if scientific contributions were to focus explicitly on the issues of greatest concern to citizens. In addition to VEC-focused assessments, Beanlands and Duinker (1983) called for greater participation from the scientific research community during the early stages of assessment planning and design. In short, Beanlands and Duinker (1983) outlined a framework for designing and implementing collaborative research programs aimed at providing useful and defensible insights into stakeholder-relevant issues.

The literature of the 1990s and 2000s has continued to emphasize the importance of good scoping in determining both the quality of EAs and the overall acceptability of EA-related development decisions (e.g., Wood 1995; Jones 1999; Treweek 1999). Still, scoping has been identified as an ongoing area of weakness in EA practice from both technical and participatory perspectives (e.g., Wood 1995; Sadler 1996; Weston 2000; Ross et al. 2006; Snell and Cowell 2006; Wood et al. 2006). Despite widespread agreement on a basic definition for scoping, two competing approaches to scoping seem to have emerged in the literature. For some (e.g., Kennedy and Ross 1992; Ross et al. 2006), scoping functions as a sort of 'reversible funnel' for adding ('scoping in') and then removing ('scoping out') issues and VECs from a list of things to address during an assessment. For others (e.g., Mulvihill and Jacobs 1998; Mulvihill and Baker 2001; Mulvihill 2003), scoping functions as a creative and inclusive design process aimed at outlining a well-defined, stakeholder-relevant framework for investigation and evaluation. Indeed, scoping may be conducted under more formal and adversarial arrangements (e.g., guidelines hearings), or under more informal and collaborative arrangements (e.g., non-adversarial stakeholder meetings, scenario-building exercises).

More recently, Morrison-Saunders et al. (2014) have argued that current perceptions of the lack of efficiency and effectiveness in EA practice can be attributed to the "proliferation of assessment types that has emerged over the past several decades". This, they argue, "is creating silos and confusion amongst regulators, stakeholders and even impact assessment practitioners; and is potentially resulting in the core principles and foundations of impact assessment being undermined". The authors go on to argue that: "the solution to the problem is to take an integrated approach which requires an emphasis in particular during the scoping stage".

Greig and Duinker (2014) were quick to challenge such conclusions, noting that: "Issues of EA scoping and integration may contribute to this problem but they are by no means the heart of it; the broader context must be understood". To this they added:

> Various other, more fundamental problems with EA process contributed to delays in assessments including shallow practice in impact prediction (Duinker and Baskerville 1986; Greig and Duinker 2011), weak approaches to significance determination (e.g., Duinker 2013), trivial assessments of cumulative effects (Duinker and Greig 2006), inadequate monitoring (Duinker 1989), and others.

Overall, Greig and Duinker (2014) concluded that

> Calling for greater effort and rigor in scoping things off any EA's agenda might help, but 30 years of doing so by Canadian scholars has led to no desirable effect whatsoever […] as long as EA is a proponent-dominated process aimed at thinking a bit about the environment while seeking development approval, none of the main problems it faces will be alleviated. EA in Canada has been a technically shallow, adversarial process that needs to become technically rigorous and collaborative.

Here I provide a review of the literature surrounding the five major themes typically addressed through EA scoping: (i) development alternatives, (ii) VECs, (iii) indicators, (iv) drivers, and (v) boundaries. I also provide a review of any literature that has helped to explain the role of science in addressing each of these five themes.

7.1.2 Alternatives

The consideration of technically and economically feasible development alternatives has been widely acknowledged as a foundational principle of EA (e.g., Andrews 1973; CEQ 1978; Beanlands and Duinker 1983; Holling 1978; Wood 1995; Sadler 1996). It has also been recognized as an important opportunity to integrate EA processes with the often separate enterprise of development planning and design (e.g., Brown and Hill 1995; McDonald and Brown 1995). Despite being a formally stated requirement in most EA laws and policies, the identification and evaluation of alternatives has been widely recognized as a major area of weakness in EA practice (e.g., Hill and Ortolano 1978; Wood 1995; Valve 1999; Steinemann 2001). While development alternatives are often described briefly at the beginning of assessments, they are rarely carried through to the impact prediction, significance evaluation, and review stages of the EA process. Instead, alternative developments are typically dismissed early on, with little accompanying rationale to justify the selection of a preferred alternative.

The premise behind the consideration of alternatives in EA is that, in general, the stated goal or purpose of a proposed development can often be achieved in different ways, each of which will have different environmental impacts. Depending on the nature of particular development proposals, four general types of alternatives may be defined at the outset of EA processes:

(i) the so-called ‘no-action’ alternative
(ii) alternative sites or locations for development
(iii) alternative development designs (i.e., ‘alternative means’ of implementing the proposed development)
(iv) functionally distinct ways of achieving development goals (i.e., ‘alternatives to’ the proposed development)

From a scientific perspective, the consideration of alternatives in EA allows for the identification of development configurations that will ensure the greatest protection of VECs. From a political perspective, the participatory design of alternatives enhances the likelihood of negotiating a mutually-agreeable decision following EA report submission and review. In short, the early identification of alternatives to be compared through impact prediction, evaluation, and review is an important element of a strong scoping process.

7.1.3 VECs

Given its ultimate purpose of securing sustainable development by protecting important environmental values (Sadler 1996), the EA process must begin by explicitly identifying VECs based on stakeholder concerns (Beanlands and Duinker 1983). Broadly speaking, VECs are specified entities of the environment (e.g., particular species, specific ecosystems, water and air quality) that are valued by members of society but potentially compromised by proposed developments. In order for VECs to accurately embody the range of environmental values held by different stakeholder groups, they must be explicitly selected on the basis of public concern, professional concern, or both. According to Beanlands and Duinker (1983), while there is no sure way of anticipating the specific environmental concerns of stakeholders, human societies in general can be expected to place a high value on: (i) environmental components related to human health and safety, (ii) biotic species of major commercial, recreational, or aesthetic importance, (iii) rare or endangered species, and (iv) suitable habitat for such species. To be sure, scientific contributions to EA will be far more useful to all participants if they focus explicitly on stakeholder-relevant VECs as well as the key ecosystem components and processes that sustain such VECs.

More recently, the notion of EA as being focused on more traditional components of ecosystems has been expanded. With growing concerns over the consequences of global climate change and biodiversity loss, such issues have been increasingly considered in the context of formal EA. While the issue of climate change is fairly specific, that of biodiversity loss implies the consideration of many biological entities operating at multiple levels of organization. Indeed, the literature (e.g., Nelson and Serafin 1991; Gontier et al. 2006) indicates that the concept of biodiversity—as with that of the ecosystem—can be dissected into components that are both readily identifiable and valued by stakeholders and decision-makers alike. Gontier et al. (2006), for example, refer to such entities as 'valued biodiversity components'.

7.1.4 Indicators

The term 'indicator' has been widely used in both ecology and environmental planning circles for some time (e.g., Moore 1966). Generally, the term has been used to mean any perceptible attribute of the biophysical environment that is to be measured repeatedly to detect environmental changes. Whereas some ecologists (e.g., Swank and Douglass 1975; Jackson et al. 1977) tended to focus on biogeochemical indicators (e.g., nutrient flux) that would reflect the aggregate condition of whole ecosystems, others (e.g., Cairns 1974; Bauerle et al. 1975; Ray and White 1976) tended to focus on biological indicators (e.g., 'indicator species', community structure) that would reflect the condition of specific ecosystem components (e.g., air and water quality).

Holling's (1978) treatise on AEAM outlined a collaborative approach to indicator selection, whereby scientists and decision-makers work together to identify measurable environmental attributes that reflect the interests and concerns of decision-makers. According to Holling (1978), indicators are "Measures of system behavior in terms of meaningful and perceptible attributes". To this he added: "Appropriate indicators for evaluation are readily generated in any assessment problem, provided that an essential constraint is understood: there is no "comprehensive" list of indicators, and there is no "right" set of indicators for any problem, ever". According to Holling (1978) the bounding (i.e., scoping) stage of EA processes was to be focused primarily on the identification of: (i) state variables, and (ii) indicators. In Holling's (1978) view, state variables represent the key components necessary to capture the behaviour of complex ecosystems, while indicators constitute the terms in which environmental effects (predicted or measured) are to be evaluated by decision-makers. Simply put, a complex problem may involve many interacting variables, but such complex information may be more easily understood and accepted by decision-makers when it can be reduced to a few key measures of performance (i.e., indicators).

In the 1980s, a growing body of peer-reviewed EA literature began using the term 'indicator' in a variety of different ways. While some authors (e.g., Rosenberg et al. 1986) continued to emphasize the importance of monitoring stressor-sensitive species (i.e., 'indicator species') to detect environment changes, others (e.g., Duinker and Baskerville 1986) emphasized the importance of identifying stakeholder-relevant ecosystem components early-on in order to facilitate environmental decision-making processes. Duinker (1989), for example, suggested that the concept of the 'VEC' outlined by Beanlands and Duinker (1983) was virtually synonymous with that of the 'performance indicator' outlined by Holling (1978). From Duinker's (1989) perspective, therefore, VECs themselves could be seen as indicators of ecosystem performance.

Decades later, Heink and Kowarik (2010) have observed confusion surrounding the term 'indicator' and its precise meaning in the context of environmental decision-making. The authors write that "The term ''indicator'' is frequently used at the interface between science and policy. Although there is a great demand for clear definitions of technical terms in science and policy, the meaning of indicator is still ambiguous". Overall, the authors recommend distinguishing between indicators: (i) as ecological components (i.e., ecological units, structures, or processes) and as measures (i.e., properties of a phenomenon, body, or substance to which a magnitude can be assigned); (ii) as descriptive and normative (i.e., prescriptive) statements about ecological conditions; (iii) as direct and indirect measures of a biophysical entity of interest; (iv) as simple indicators and complex (i.e., composite) indices; and (v) as measures of simple ecosystem components (e.g., wildlife populations) and measures of more complex ecosystem components (e.g., biodiversity).

In my view, VECs (including both simple and complex ecological entities) embody the values and concerns of stakeholders and decision-makers. Indicators, on the other hand, constitute the terms in which environmental changes are predicted, communicated, and subsequently measured (Duinker et al. 2013). Though some authors have favoured the use of aggregate indices of environmental impact (e.g., Dee et al. 1973; Antunes et al. 2001; Canter and Atkinson 2011), I note that the concept of indicators is much more basic. Indicators, simply put, are measurable attributes of the environment that describe the state or condition of VECs (Duinker et al. 2013). While an individual VEC may have more than one indicator attached to it, direct measurement of VEC condition is always preferred over indirect measurement. In some instances, where direct measurement of VEC condition is not feasible (e.g., rare species population), indirect measurement of a related environmental attribute may suffice (e.g., suitable habitat). In such cases, the relationship between the selected indicator and the VEC condition being measured ought to be clearly established and periodically verified.

7.1.5 Boundaries

The importance of setting reasonable time and space boundaries for EAs has been widely acknowledged in the literature (e.g., Holling 1978; Duinker and Baskerville 1986; Clark 1994; Noble 2000; João 2002; Geneletti 2006; Karstens et al. 2007). In general, it is argued that the explicit delineation of study boundaries in space and time provides a further level of specificity and focus to the design and conduct of EAs.

Beanlands and Duinker (1983) wrote that the delimitation of EA studies in space and time requires explicit consideration of multiple overlapping human-imposed and naturally occurring boundaries. These include jurisdictional and study-length limitations imposed by administrative authorities, the spatial extent and lifespan of proposed developments, and the temporal and spatial scales at which ecosystems naturally operate. Within the limits imposed by administrative authorities, Beanlands and Duinker (1983) argued that participants should begin by first considering an initial set of spatial boundaries based on physical transport mechanisms (e.g., movement of water and nutrients). Because physical boundaries do not always match biological ones, Beanlands and Duinker (1983) argued that an initial consideration of physical dynamics in space should be followed by a consideration of biological dynamics in space (e.g., animal migration). In the words of Beanlands and Duinker (1983): “Although not universally accepted, the principle of setting physical boundaries first, followed by biological bounding, was stressed in many of the workshops”.

Beanlands and Duinker (1983) observed that reasonable study boundaries in time could be established on the basis of a variety of temporal characteristics of ecosystems. According to Beanlands and Duinker (1983), "Such factors include: (i) the magnitude, periodicity and trends in the natural variation of the variables of interest, (ii) the time required for a biotic response to become evident, and (iii) the time required for a system or subsystem to recover from a perturbation to its pre-impact state". Beanlands and Duinker (1983) further likened such temporal ecological boundaries to the concept of 'resilience' outlined by Holling (1973), as well as the concept of 'recoverability' outlined by some other authors (e.g., Cairns 1980; Cairns and Dickson 1980). Overall, Beanlands and Duinker (1983) concluded that although conceptually appealing, such theoretical notions had yet to find practical application in the detection of temporal ecosystem boundaries. In the words of Beanlands and Duinker (1983): "Unfortunately, most of the research concerning stability or resiliency within natural systems […] has progressed little beyond the conceptual or theoretical stage with limited direct application to determining boundaries in environmental impact assessment".

More recently, some authors (e.g., Clark 1994; João 2002; Geneletti 2006) have continued to explore the task of boundary-setting from a technical or scientific perspective, while others (e.g., Karstens et al. 2007) have begun to examine the issue from a participatory perspective. In my view, collaborative boundary-setting through scoping continues to be an important aspect of focusing EA studies. Since the Beanlands and Duinker (1983) report was published, however, very few guidance materials have emerged on the subject of boundary-setting in EA. In the words of Noble (2000): "Defining the boundary within which to conduct an environmental impact assessment is often a challenging task. Perhaps the area of greatest concern, and ironically the area of least attention, is the definition of temporal boundaries in EIA". Perhaps the setting of impact study boundaries will always be a challenge due to irreducible uncertainties surrounding the spatial and temporal dynamics of complex ecosystems, particularly in the context of a changing global climate. Therefore, as with other elements of ecosystems identified during the scoping process, it may be best to consider initial study boundaries as being tentative. In this way, uncertain or fluctuating ecosystem boundaries may be iteratively adapted throughout the EA process.

7.1.6 Drivers

Holling (1978) wrote that "In choosing variables we must be careful to distinguish between system state variables and driving variables". According to Holling (1978), state variables represent the ecological quantities we wish to predict, whereas as driving variables represent quantities known to influence the condition of state variables, but for which the model does not make predictions. In other words, predictions about the future condition of system state variables are based largely on assumptions about the behaviour of system driving variables. Moreover, driving variables may represent either environmental phenomena operating external to the ecosystem in question, or development actions imposed by humans onto that ecosystem. In the words of Holling (1978):

> One way to define a driving variable is to say that it is some factor whose variation is determined by forces outside the arbitrary boundaries of the system under study, e.g., light conditions. When we change a model to include calculations or predictions about a factor that we have previously called a driving variable, then that factor is no longer called a driving variable but is instead part of the arbitrary system (a system or state variable).

Holling (1978) indicated that one way to handle uncertainty in drivers (not to mention relationships and parameters) would be to construct what he called alternative models. According to Holling (1978), such models would allow for the inclusion of potentially influential (yet highly uncertain) environmental factors. With respect to building alternative models, Holling (1978) concluded that:

> Clearly, one of the most valued and effective traits a manager can possess is his ability to see (and therefore to model) a problem from a wide range of perspectives. In practice, most interpretations (i.e., models) offered for a problem tend to be shaped by habitual ways of thinking, and effective "new looks" are

> most difficult to establish. Consensus-breeding techniques are your enemy in this situation, and imagination is your only sure friend.

Decades later, the issue of uncertainty surrounding external drivers, particularly global climate, has spurred on new conversations about the handling of uncertainty during the scoping stage of EA processes. More specifically, some authors (e.g., Mulvihill 2003; Duinker and Greig 2007; Bond et al. 2015) have called for greater use of scenarios and scenario building to accommodate uncertainties in ecosystem drivers (e.g., climate) as well as other external wildcards (e.g., induced development, market demand). According to Mulvihill (2003), the use of such techniques necessarily opens up the scoping process to consider a wider range of speculative inputs, particularly regarding which contextual factors may be considered influential on future environmental conditions. According to Bond et al. (2015), scoping in EA should seek to "identify drivers of change and alternative system response trajectories". To this they added: "this exercise needs to focus on scenario building […], with stakeholders aiming to identify, as well as internal influences, the key external influences that will have an effect on the socio-ecological system, but which cannot themselves be controlled". Similarly, Duinker and Greig (2007) wrote that "given uncertainty about the future conditions (natural system drivers and patterns of human development) that will come to affect VECs over the life time of many developments, EIA predictions of the future effects within a most likely future have a high potential to be really wrong". To deal with the issue of uncertainty in external variables, Duinker and Greig (2007) concluded: "We need to include in the set of an EIA's scenarios a comprehensive range of potential future developments, and all the key driving forces, such as climate change and human demographics, that can measurably affect the VECs".

7.2 ECOLOGICAL CHARACTERIZATION

7.2.1 Overview

Foundational guidance materials for science in EA (e.g., Holling 1978; Ward 1978; Munn 1979; Beanlands and Duinker 1983; Duinker and Baskerville 1986) emphasized the importance of establishing explicit, quantitative descriptions of ecological processes as a basis for subsequent experimentation, prediction, and monitoring efforts. According to this early literature, a process-based modelling exercise offers the following special benefits to EA: (i) synthesis of ecological knowledge, (ii) transparency of assumptions, and (iii) a framework to guide empirical studies. Holling (1978) observed that once the underlying functional form of an ecological relationship had been established, additional information would be required for the estimation of: (i) parameters, (ii) initial state variables, and (iii) driving variables. For my purposes, the descriptive (i.e., characterization) stage of EA is aimed at obtaining, interpreting, and synthesizing such information.

The earliest frameworks for conducting EAs (e.g., Leopold et al. 1971; Dee et al. 1973) could depict neither ecological relationships nor spatio-temporal dimensions. Within such static and compartmentalized study frameworks, the 'count everything' approach to description inevitably prevailed. Indeed, early EAs were typically characterized by the use of a one-time, comprehensive survey to identify and list all biophysical entities observed within a given study area. Subsequent commentaries and guidance materials (e.g., Holling 1978; Ward 1978; Munn 1979; Hilborn and Walters 1981) challenged EA's preoccupation with comprehensive surveys and inventories. Ward (1978), for example, criticized two common approaches to environmental description which she labelled the "busy taxonomist" approach and the "information broker" approach. Similarly, Hilborn and Walters (1981) referred to traditional baseline and process studies in EA as "helicopter ecology". Overall, they concluded that it was not exhaustive inventories or baselines that provided the key to understanding ecological systems, but the qualitative insight of a few experts into how ecosystem components and processes interact over space and time.

Holling (1978) pointed out that a more effective and efficient approach to characterizing ecosystems would be to establish functional relationships based on existing ecological knowledge. According to Holling (1978), “one can proceed farther than is normally thought possible in the face of meager data by mobilizing available insight into the system’s constituent processes.” Holling (1978) further stated that “as soon as we know that a particular mathematical function will describe a process, the information requirements are suddenly reduced greatly. Now we need only estimate values for the few parameters of that function.” On the subject of description, Holling (1978) concluded that “the goal, then, of description is not description but useful explanation”.

In outlining the fundamental purpose of effects monitoring in EA, Duinker (1989) challenged the perceived need for so-called ‘baseline’ studies, an often time-consuming and resource-intensive endeavour to produce exhaustive descriptions of environmental conditions prior to development. Duinker (1989) reasoned that the most common approach to defining an impact—a comparison of ‘before’ and ‘after’ measurements of environmental conditions—is based on the false assumption that such conditions are static (i.e., that they would have persisted in the absence of development). Likewise, he reasoned that a comparison of measurements taken in the ecosystem of interest with measurements taken in a similar ecosystem—another common approach to defining an impact—is based on the false assumption that both ecosystems are identical, and that one would have behaved exactly like the other in the absence of development.

Duinker (1989) concluded that the most logical approach to defining an impact would be to calculate the difference between two predicted time-series: one with the proposed development in place and one without. A time-series of data generated through monitoring would then serve as a check on one of the predicted time-series. Accordingly, Duinker (1989) argued that a statistically rigorous description of natural variability (i.e., ‘baseline’)—such as that advocated by Hirsch (1980) and Beanlands and Duinker (1983)—could only modestly assist in the process of impact prediction. He argued that because natural variation in the condition of a VEC is typically caused by several ecological processes, analysts should strive to understand how such processes contribute to the observed pattern of variability, rather than simply describe that pattern.

The literature of the 1990s and 2000s has not challenged the basic tenets of ecological characterization as described in the literature of the 1980s. Since that time, however, technological advancements in computational modelling (e.g., individual-based models) and remote/automated data acquisition (e.g., LIDAR, GPS telemetry) have significantly expanded the range of tools, techniques, and datasets available for ecological characterization (e.g., Grimm 1999; Cooke et al. 2004; de Leeuw et al. 2010). The potential contributions of other kinds of ecological knowledge (e.g., local, traditional, Aboriginal) have also been the subject of considerable attention in the recent literature (e.g., Stevenson 1996; Usher 2000). In this section, I chart the evolution of the major tools and techniques available for ecological characterization in EA.

7.2.2 Mobilizing Science Outside EA

7.2.2.1 Functional Relationships

At the time of EA's inception in the early 1970s, ecological simulation modelling was just beginning to flourish. According to Jørgensen (2008), the earliest approaches to modelling represented three broad domains of ecology: (i) biogeochemical (i.e., flows of matter through ecosystems), (ii) bioenergetics (i.e., flows of energy through ecosystems), and (iii) population/community (i.e., growth and dynamics of wildlife populations/communities). The early literature on ecological modelling (e.g., May 1973; Maynard Smith 1974; Pielou 1977) further subdivided such models based on the following characteristics: (i) state-dependent or time-dependent, (ii) continuous or discrete, and (iii) stochastic or deterministic. Additionally, population models could be assembled with or without age and class structure.

Early 'compartment' or 'food web'-type models were designed to simulate flows of carbon, energy, and nutrients through different kinds of ecosystems, including grasslands (e.g., Bledsoe et al. 1971; Patten 1971; Anway et al. 1972; Innis 1975), forests (e.g., Shugart et al. 1974; O'Neill 1975; Overton 1975), deserts (e.g., Goodall 1975), tundra (e.g., Miller et al. 1975), and lakes (e.g., Walters and Efford 1972; Park 1974).

Early population simulation models were designed to simulate a range of predator-prey or animal-habitat interactions. Examples included a model for simulating the dynamics of a caribou-lichen system (Walters et al. 1975), a model for simulating the dynamics of a wolf-moose system (Zarnoch and Turner 1974), and a model for simulating the dynamics of a lamprey-trout system (Lett et al. 1975).

Early community simulation models included the so-called forest 'gap' models (e.g., Botkin et al. 1972; Ek and Monserud 1974; Shugart and West 1977), designed to simulate the successional dynamics of small forest stands following an opening in the canopy.

The 1970s also saw the widespread use of hydrological models, which were designed to simulate the physical and chemical dynamics of ground water (e.g., Freeze 1971; Winter 1978) and surface water (e.g., Quick and Pipes 1976; Solomon and Gupta 1977). Likewise, a range of geomorphological models were designed to simulate the dynamics of erosion and sedimentation processes (e.g., Onstad and Foster 1975; Bridge and Leeder 1979).

Finally, the 1970s saw the application of general systems theory to ecological modelling, which aimed to integrate the various sub-disciplines outlined above. Walters (1974), for example, described an interdisciplinary approach to watershed modelling, whereby hydrodynamics, vegetation growth, and a variety of fish and wildlife populations were functionally linked together.

At the time of the Beanlands and Duinker (1983) report, scientists were beginning to focus more explicitly on the spatial dimension of complex ecosystems, as evidenced by the emerging field of landscape ecology (e.g., Forman and Godron 1981; Wiens et al. 1985). According to the literature (e.g., Huston et al. 1988; Baker 1989; Sklar and Constanza 1991), advances in computational power, coupled with the advent of GIS and satellite remote sensing, allowed for the development of individual-based and spatially explicit simulation models. In particular, Huston (1988) argued that such models would foster greater collaboration and synthesis among the various scientific disciplines. Early examples included a model for simulating hydrological flows and vegetation succession in a forested floodplain (Pearlstine et al. 1985), a model for simulating the physical transportation of water and sediments in a coastal marsh (Costanza et al. 1988; Sklar et al. 1985), and a model for simulating fire and post-fire succession in wet or semi-arid forests (Kessel et al. 1984).

With growing interest in Earth's global biogeochemical cycles, the 1980s also saw the development of numerous models for linking cross-scale atmospheric, marine, and terrestrial processes. Examples included models for simulating the biogeochemical and successional dynamics of coupled soil-vegetation systems (e.g., Solomon 1986; Pastor and Post 1988; Urban and Shugart 1989), models for simulating the biogeochemical and climatic dynamics of coupled vegetation-atmosphere systems (e.g., Dickinson and Henderson-Sellers 1988; Sellers et al. 1986; Running and Coughlan 1988; Hall et al. 1988), and models for simulating the biogeochemical, hydrological, and successional dynamics of forests, lakes, and watersheds under acid precipitation regimes (e.g., Booty and Kramer 1984; Cosby et al. 1985; Alcamo et al. 1987; Nikolaidis et al. 1988).

The literature of the 1990s and 2000s has since revealed widespread interest in the development of individual-based and spatially explicit simulation models. For example, a number of spatially explicit landscape models have been developed for simulating the productivity and successional dynamics of broad-scale forest landscapes (e.g., Mladenoff and He 1999; Scheller and Mladenoff 2004; Scheller et al. 2007). Likewise, a number of spatially explicit population models have been developed for simulating the dispersal and survival of wildlife populations in 'patchy' landscapes (e.g., Mckelvey et al. 1992; Pulliam et al. 1992; Dunning et al. 1995). Similar models have also been developed for freshwater fish populations (e.g., Van Winkle et al. 1998).

In order to incorporate more realistic habitat dynamics, a number of authors have proposed frameworks for linking spatially explicit population models with forest landscape models using habitat suitability indices (e.g., Akçakaya 2001; Larson et al. 2004; McRae et al. 2008; Franklin 2010). Such approaches to modelling, however, have been controversial, as the rules governing animal dispersal, movement, and mortality remain highly sensitive to parameter adjustments (see Ruckelshaus et al. 1997; Mooij and DeAngelis 1999). Moreover, other authors (e.g., Morrison et al. 2006) have cautioned that such models do not consider important processes such as predation and interspecific competition, which can have a strong influence on population dynamics. Still other authors (e.g., Railsback and Harvey et al. 2002) have observed that the simplistic rules used to simulate animal movement often fail to reproduce realistic patterns of behaviour.

Indeed, more defensible, systems analytical approaches to simulation modelling seem to have had lasting appeal in the scientific literature. For terrestrial wildlife, a number of population models have been developed to simulate the dynamics of predator-prey-habitat systems (e.g., Weclaw and Hudson 2004). For marine life, fisheries scientists have found it more useful to rely on population models linked to 'trophic compartment'-type models (e.g., McClanahan 1995; Walters et al. 1999; Christensen and Walters 2004). This, together with ongoing interest in modelling the carbon, water, and nutrient budgets of terrestrial ecosystems (e.g., Arnold and Allen 1996; Billen and Garnier 1999; Kurz et al. 2009), suggest that perhaps not much has changed in the fundamentals of ecological modelling since the 1980s.

7.2.2.2 Parameters and Process Studies

According to the scientific literature (e.g., Richter and Sondergrath 1990; Hilborn and Walters 1992; McCallum 2000), parameters represent a crucial link between data and models, or between empirical and theoretical science. It is also widely recognized (e.g., Jørgensen and Bendoricchio 2001; Soetart and Herman 2009) that the most consistent and reliable parameter estimates are derived from the published results of long-term empirical studies. Broadly speaking, different kinds of models will require different kinds of parameters, but the following general types can be delineated: (i) life-history (e.g., Hindell 1991; Wich et al. 2004), (ii) demographic (e.g., Fancy et al. 1994; Best et al. 2001), (iii) physiological (e.g., Brett and Glass 1973; Sullivan et al. 1996), (iv) biogeochemical (e.g., Soer 1980; Kavvadias et al. 2001), (v) hydrological (e.g., Reimers 1990; Zavattaro and Grignani 2001), and (vi) disturbance (e.g., Anderson et al. 1987; Seymour et al. 2002). Many helpful guides to parameterizing and calibrating simulation models can be found in the literature (e.g., Janssen and Heuberger 1995; Jørgensen and Bendoricchio 2001). Here I chart the general development of empirical biophysical science since the 1970s, as described in the formal peer-reviewed literature. In doing so, I attempt to highlight major technological and methodological advancements related to the study of ecological processes and parameters.

At the time of EA's inception in the early 1970s, ecological research was dominated by short time frames and a focus on local-scale sites (CEQ 1974; Callahan 1984). For wildlife studies, many scientists (e.g., Dolbeer and Clark 1975; Wilbur 1975) relied on the use of traditional mark-recapture and trapping techniques to estimate demographic parameters, while other scientists (e.g., Cook et al. 1971; Carroll and Brown 1977) had begun to use VHF radio telemetry. In addition to population studies, radio telemetry could be used to characterize animal home ranges (e.g., Van Ballenberghe and Peek 1971; Trent and Rongstad 1974), habitat use (e.g., Craighead and Craighead 1972; Wallestad 1971; Nicholls and Warner 1972), and physiological state (e.g., Priede and Young 1977; MacArthur et al. 1979).

To characterize biogeochemical flows of matter and energy between ecosystem 'compartments', many scientists (e.g., Schlesinger 1977; Whittaker et al. 1979) relied on traditional field and laboratory techniques such as isotopic labelling, chromatography, spectrophotometry, elemental analyzers, and gas exchange methods. Meanwhile, other scientists (e.g., Kelly et al. 1974) had begun to profit from the use of automated, in-situ measurement devices such as electrochemical sensors.

In terms of broad-scale, collaborative research, the 1970s saw the end of the International Biological Program (IBP), a decade-long, multi-site research effort focused on the study of ecological productivity (NAS 1975). Despite its many shortcomings, the IBP would make a number of important contributions to the study of biogeochemical processes, particularly with respect to soil nutrients, organic decomposition, and plant productivity (e.g., Gozs et al. 1973; Whittaker and Likens 1975; Whittaker et al. 1979).

Finally, the 1970s saw the launch of the first land remote sensing satellites (Landsat 1-3), whose continuous stream of data would be used, amongst other things, to characterize the photosynthetic properties of tree canopies (e.g., Rouse et al. 1973; Tucker 1979). Indeed, the initiation of the Landsat program in the 1970s marked the beginning of a new era in global, space-based monitoring systems.

In the 1980s, the field of ecology expanded to consider functional processes operating at diverse spatial and temporal scales (e.g., Holling 1986). To foster greater continuity and collaboration in ecological research, the scientific and policy communities came together in 1980 to establish the Long-Term Ecological Research (LTER) network, a collaborative, multi-site research program based on standardized methods of observation and data storage (e.g., Likens 1983; Callahan 1984, Webster et al. 1985; Strayer et al. 1986). Throughout the 1980s, the LTER program would begin characterizing a range of ecological processes and parameters within different biomes, with a particular emphasis on nutrient cycling, primary productivity, and forest succession (e.g., Knapp and Seastedt 1986; Mattson et al. 1987; Swanson et al. 1988; Burke et al. 1989; Magnuson et al. 1990; Aber et al. 1990).

To advance the study of Earth's global biogeochemical systems, and to gain a better understanding of the role of humans in altering those systems, the international scientific community came together in 1980 to establish the World Climate Research Program (WCRP), and again in 1986 to establish the International Geosphere Biosphere Program (IGBP) (e.g., Schiffer and Rossow 1983; NRC 1986b). Throughout the 1980s, the WCRP and IGBP would together rely on space-based remote sensing, in coordination with ground-based monitoring efforts (e.g., LTER), to characterize a range of cross-scale biogeochemical processes linking Earth's atmosphere, oceans, and land surfaces (e.g., Hall et al. 1988; Sellers et al. 1988; Goward and Hope 1989; Houghton et al. 1990).

In a related set of developments (e.g., NASA 1984), significant improvements were made in high resolution and multi-spectral remote sensing (e.g., NOAA-AVHRR, Landsat-TM). These new instruments, along with improved analytical techniques, would allow for the estimation of more-accurate model parameters, particularly those associated with tree canopy photosynthesis and productivity (e.g., Holben 1986; Rock et al. 1986; Huete 1988).

Finally, the late 1980s saw the application of Argos—a satellite-based weather monitoring system—to the tracking of free-ranging wildlife (Fancy 1988). Unlike traditional radio transmitters, which were limited by their short signal range, Argos-based systems were capable of tracking animal migrations over vast distances (e.g., Strikwerda et al. 1986; Fancy et al. 1989). Moreover, by coupling these data with simultaneous remote sensing measurements in a GIS, broad-scale animal movements could be linked to dynamic environmental conditions and habitat characteristics (e.g., Priede 1984).

The literature of the 1990s and 2000s has since highlighted many new technologies with potential applications in biophysical science. New satellite-borne remote sensing instruments (e.g., MODIS) offering greater spectral coverage and resolution have provided new and improved datasets for use in ecological studies (Justice et al. 1998). Noteworthy applications include the characterization of wildfire regimes (Justice et al. 2002), enhanced vegetation indices (Huete et al. 2002), and global land cover mapping (Friedl et al. 2002). Recently, aircraft-borne LIDAR sensors have emerged as a viable option for characterizing the distribution, three-dimensional structure, and functional characteristics of forest stands (Lefsky et al. 2002).

Likewise, a number of technological advancements in satellite telemetry (e.g., GPS-based systems, miniaturized tags, coded signals, lithium batteries, archival loggers) have allowed for more-reliable and more-accurate animal tracking studies (Rodgers 2001; Cooke et al. 2013). Additionally, devices that provide information on an animal's physiology, behaviour, and energetic status have recently emerged as a viable option for characterizing interactions between individual animals and their environment (Cooke et al. 2004). Finally, the 1990s and 2000s have seen the establishment of automated, wireless sensor networks for monitoring a variety of biogeochemical parameters, fluxes, and state variables at a variety of spatial and temporal scales (Collins et al. 2006; Hart and Martinez 2006; Jones et al. 2010). When simultaneously integrated with adjacent sensor networks, remote sensing platforms, and GPS/Argos telemetry systems, these multi-tiered monitoring platforms have the potential to generate powerful insights into the relationships linking local animal behaviour, regional environmental conditions, and global-scale biogeochemical drivers (Rundel et al. 2006; Handcock et al. 2009).

In summary, empirical biophysical science has changed significantly since the time of EA's inception in the early 1970s. Driven primarily by advances in theory and technology, local-scale biophysical science has been progressively expanded to regional and global scales through integration with Earth systems science. The long-term monitoring efforts associated with these developments have resulted in more robust characterizations of ecological parameters, as well as a greater understanding of the dynamic processes and feedbacks linking global climatic and biogeochemical changes to local-scale ecosystems, communities, and populations.

7.2.3 Data Collection Inside EA

7.2.3.1 Field Surveys

While long-term ecological research is thought to provide the most consistent and reliable parameter estimates, site-specific studies are still needed inside EA to characterize the initial conditions of VECs, other state variables, and driving variables. Considering the limited time frames in which most EAs must operate, Holling (1978) emphasized the importance of accessing existing inventories and databases wherever possible, including those maintained by government agencies and industry groups. Ward (1978), on the other hand, outlined some basic approaches for designing and implementing short-term field surveys in EA. In general, she described the use of mark-recapture techniques to estimate wildlife abundances, and the use of field sampling and analytical chemistry techniques (e.g., elemental analyzers, spectrophotometry, chromatography) to characterize physical media such as water and soils. Subsequent guidance materials published in the 1980s (e.g., Beanlands and Duinker 1983; Westman 1985; NRC 1986a) described the use of newer tools and techniques (e.g., satellite remote sensing, radio telemetry) to characterize important wildlife habitat.

Decades later, the subject of field surveys has not received much attention in the EA literature. A quick search of the scientific literature, however, reveals several detailed protocols for the characterization of air (e.g., Chow 1995; Baron and Willeke 2001), water (e.g., Hounslow 1995; Fetter 2000), soils (e.g., Carter and Gregorich 2008), terrestrial carbon stocks (e.g., Brown 2002; Gibbs et al. 2007), wildlife populations (e.g., Seber 1982; Buckland et al. 2015), and wildlife habitats (e.g., Manly et al. 2002).

7.2.3.2 Local, Traditional, and Aboriginal Knowledge

Recently, the potential application of other kinds of ecological knowledge (local, traditional, Aboriginal) has been the subject of considerable discussion in the literature (e.g., Freeman 1992; Berkes 1993; Johannes 1993; Sallenave 1994; Stevenson 1996; Huntington 2000; Usher 2000). In general, these authors delineate some major categories of informal ecological knowledge, the overall applicability of such knowledge to EA processes, and some of the major challenges associated with its collection, interpretation, and integration with formal scientific knowledge. According to Stevenson (1996), many Aboriginal resource users have had extensive contact with their local environment, and therefore possess a richer and more detailed understanding of it than most outsiders do. Stevenson (1996) explained that based on their familiarity with local environments, Aboriginal resource users can, if they so choose, contribute valuable insights to EA studies, particularly with respect to wildlife and wildlife habitat. In a similar vein, Usher (2000) noted that the substantial time-depth of traditional observations promises to provide better explanations for existing environmental conditions than descriptions derived from short-term field surveys, particularly in remote and poorly studied ecosystems. Both Stevenson (1996) and Usher (2000) pointed out that traditional ecological knowledge studies often require far less time and money to conduct than conventional baseline studies.

Overall, both authors concluded that for traditional and Aboriginal knowledge to realize its full potential in EA, knowledgeable elders and resource users must be encouraged to play an active role in determining how their knowledge will be documented, interpreted, and used within the EA process. In absence of such collaborative arrangements, there is a distinct risk that such knowledge may be misinterpreted, misused, and even exploited by non-Aboriginal interests.

7.2.3.3 Integrating Field Surveys and Traditional Knowledge

The recent literature provides numerous examples of how scientific and traditional observations can be integrated to explain complex ecological phenomena, particularly in northern and other remote environments (e.g., Huntington et al. 2004a, 2004b; Gilchrist et al. 2006; Gagnon and Berteaux 2009). According to Gagnon and Berteaux (2009), traditional ecological knowledge "has generally been recognized as differing from science because it is based on information acquired during longer time series but over smaller and more specific localities." Considering their differences, Huntington et al. (2004a) observed that both kinds of knowledge may be mutually exchanged and integrated in order to understand complex, multi-scale phenomena.

Summarizing recent collaborations with resource users in the Arctic, Huntington et al. (2004b) explained how satellite telemetry data describing the broad-scale migration patterns of duck and whale populations were paired with the traditional ecological knowledge of local hunting communities to explain recently observed population declines. In one example, telemetry data showed that ducks breeding in the Northwest Territories during the summer months migrated each year to nesting and moulting areas in Alaska during the winter months. Huntington et al. (2004b) concluded that the timing and migratory connection between the two distant areas suggests that locally observed population declines in the Northwest Territories could be attributed to a viral disease outbreak in Alaska that caused the collapse of a local herring population (an important food source for migratory waterfowl).

In a similar example, Gilchrist et al. (2006) showed how local and formal scientific knowledge could be integrated to explain locally observed duck population declines in Nunavut. They conclude that the timing and circumstances surrounding a particular mass die-off event suggest links between a volcanic eruption in the Philippines, subsequent atmospheric cooling in the Polar Regions, and persistent winter ice that could prevent feeding by the ducks in open water polynyas. Such case studies provide strong evidence for the integration of traditional and formal scientific knowledge inside EA, particularly when developments are proposed in remote and poorly studied ecosystems.

7.3 Cause-Effect Research

7.3.1 Need for Cause-Effect Knowledge Inside EA

Holling (1978) maintained that long-term observational studies, though they provide the data needed to establish functional process relationships and initial parameter estimates, only reflect the behaviour of ecological systems within a narrow range of historical or naturally occurring conditions. Holling (1978) reasoned that because ecosystems typically display complexities in their response to disturbance events (e.g., nonlinearities, time lags, thresholds), extrapolation beyond observed conditions cannot reliably predict the effects of novel development actions. Consequently, he and others (e.g., Ward 1978; Hilborn and Walters 1981; Suter 1982; Beanlands and Duinker 1983; NRC 1986a; Walters and Holling 1990; Walters 1993) advocated the use of carefully designed laboratory, field, and ecosystem experiments to reduce uncertainty surrounding the effects of particular development actions.

According to Holling (1978), understanding how an ecological parameter or variable of interest (e.g., reproductive rate, survival rate) might respond to a particular development action would require the following steps: (i) disaggregation of that parameter or variable into its constituent components/processes (e.g., mating, nesting, food supply), and (ii) design of an experiment or set of experiments to study the effects of proposed development actions on each of those components. Holling (1978) also noted, however, that due to the broad spatial and temporal scales over which many development-related impacts occur (e.g., hydroelectric dams, bioaccumulation of toxins), it may be difficult or even impossible to design and conduct the necessary experiments. Accordingly, he and others (e.g., Ward 1978; Beanlands and Duinker 1983; NRC 1986a; Walters 1986) emphasized the importance of treating the development itself as an experiment. In this way, anticipated effects would be predicted prior to development, actual effects monitored during implementation, and new knowledge created to support predictive endeavours in the future.

Recently, Greig and Duinker (2011) have observed that the short time frames in which most EAs are conducted generally preclude the use of large-scale perturbation experiments, which themselves require ample time and resource to conduct. Such experimentation, they argue, is better suited to science outside EA, which is well-equipped to provide scientists inside EA with the resulting effects knowledge. To complete the circle, science inside EA must then predict and monitor the effects of development, thereby testing and refining the general knowledge provided by science outside EA. Here I chart the evolution of experimental cause-effect research conducted outside EA, as described in the scientific literature published since the early 1970s.

7.3.2 Creation of Cause-Effect Knowledge Outside EA

In general, most cause-effect experiments conducted in the 1970s were aimed at understanding the demographic, behavioural, physiological, hydrological, and biogeochemical consequences of pollution. Examples included the use of laboratory bioassays to study the effects of water pollution on fish (e.g., Leach and Thakore 1975; Hara et al. 1976; Dawson et al. 1977), field plots and fumigation chambers to study the effects of air and soil pollution on plants (e.g., Shure 1971; Hill et al. 1974; Koterba et al. 1979), and loud equipment or noise-making devices to study the effects of noise pollution on wildlife (e.g., Freddy et al. 1977; White et al. 1979). A number of small experiments were also conducted to study the effects of thermal pollution and mechanical entrainment/impingement on young fish, eggs, and larvae drifting or swimming near thermal power plants (e.g., Rulifson 1977; Dorn et al. 1979).

In addition to simple laboratory and field experiments, the 1970s saw widespread interest in modelling the effects of pollution using artificially simplified ecosystems. Examples included indoor microcosms for studying the effects of soil pollution on forests (e.g., Jackson et al. 1978; Ausmus et al. 1978), and outdoor microcosms for studying the effects of water pollution on marine biota (e.g., Evans 1977). Somewhat larger and more ambitious research initiatives relied on outdoor 'mesocosms' to simulate the effects of pollution on marine ecosystems (e.g., Menzel and Case 1977; Kremling et al. 1978; Smith et al. 1979).

Finally, the 1970s saw some of the first whole-ecosystem perturbation experiments, which were mostly aimed at understanding the causes and effects of water pollution. Examples included large-scale experiments for studying the effects of fertilization and nutrient enrichment on entire lakes (e.g., Schindler et al. 1971), and watershed experiments for studying the effects of deforestation and herbicide application on stream water (e.g., Likens et al. 1970; Douglass and Swank 1975).

In the 1980s, large ecosystem experiments expanded to accommodate growing concerns over the effects of acid rain. Examples included the artificial acidification of streams, lakes, and wetlands (e.g., Hall et al. 1980; Schindler et al. 1980; Bayley et al.

1987; Turner et al. 1987; Swenson et al. 1987), and the use of glasshouses to study the effects of acid deposition on forests (e.g., Wright et al. 1986, 1988).

In addition to ongoing pollution research, the 1980s saw some of the first large experiments aimed at characterizing the effects of species removals, species additions, and habitat disturbances. Examples included whole-lake experiments for studying the effects of fish population removals and additions on lake productivity and nutrient cycling (e.g., Carpenter et al. 1987; Carpenter and Kitchell 1988; Carpenter et al. 1988; Elser and Carpenter 1988), experiments for studying the effects of aircraft noise on wildlife habitat-use (e.g., Krausman et al. 1986), wetland experiments for studying the effects of water level manipulation on vegetation succession and waterfowl habitat selection (e.g., Murkin and Kaldec 1986; van der Valk et al. 1994; Murkin et al. 1997), and grassland experiments for studying the effects of mowing, burning, and cattle grazing on the abundance, diversity, and productivity of prairie vegetation (e.g., Abrams et al. 1986; Hulbert 1986; Abrams and Hulbert 1987; Collins 1987; Gibson 1989).

The 1980s also saw the initiation of the first large habitat fragmentation experiments. By deliberately cutting and burning swaths of tropical rainforest, such experiments would begin to understand the effects of habitat fragmentation on species diversity, population demographics, and local microclimate in remnant forest patches of various sizes (e.g., Lovejoy et al. 1986; Malcolm 1988; Rylands and Keuroghlian 1988; Harper 1989; Kapos 1989; Bierregaard and Lovejoy 1989; Bierregaard et al. 1992).

More recently, cause-effect experiments have again expanded to accommodate growing concerns over the effects of climate change. Examples include the use of glasshouses to study the effects increased temperature and CO_2 on forests (e.g., van Breemen et al. 1998; Wright 1998), the use of 'free air CO_2 enrichment' technology to study the effects of increased temperature and CO_2 on forests, grasslands, and deserts (e.g., Hendrey et al. 1999; Jordan et al. 1999; Dickson et al. 2000; Reich et al. 2001), and the use of field enclosures and mesocosms to study the effects of nutrient enrichment, acidification, and elevated CO_2 on coral reefs (e.g., Koop et al. 2001; Langdon et al. 2003; Anthony et al. 2008).

In addition to climate change concerns, a number of emerging industries and technologies have posed new and potentially unacceptable environmental consequences (e.g., transgenic crops, aquaculture, seabed mining, hydraulic fracturing). Though the environmental impacts of these emerging industries were largely unknown a few decades ago, experimental research has been successful in reducing some of the critical uncertainties. Examples include the use of laboratory, field, and watershed experiments to study the effects of hydraulic fracturing and shale gas development on water, soil, and forest vegetation (e.g., Murdoch 1992; McKay et al. 1993; Adams 2011; Adams et al. 2011; McBroom et al. 2012), the use of laboratory, field, and glasshouse experiments to study the effects of genetically modified crops on soil ecosystems (e.g., Andersen et al. 2007; Birch et al. 2007; Cortet et al. 2007), and the use of whole-lake experiments and pilot studies to examine the effects of fish farming on freshwater and marine ecosystems (e.g., Findlay et al. 2009; Rooney and Podemski 2009; Aguado-Giménez and Ruiz-Fernández 2012). Likewise, widespread concern over the increasing exploitation of seabed resources has led to a range of benthic impact experiments. Examples include the use of experimental trawling and dredging to study the effects of fishing gear and aggregate extraction on marine benthos (e.g., Kenny and Rees 1996; Hall-Spencer and Moore 2000; Bradshaw et al. 2001; Gilkinson et al. 2003; Kenchington et al. 2006), and experimental seabed disturbances to study the effects of deep sea mining on marine ecosystems (e.g., Trueblood and Ozturgut 1997; Yamazaki and Kajitani 1999; Thiel et al. 2001; Sharma 2001).

As pointed out by Greig and Duinker (2011), science outside EA has demonstrated its ability to design and implement rigorous, long-term experiments in large ecosystems. The scientific enterprise inside EA, which is often faced with severe time restrictions, could therefore benefit substantially by drawing upon such external resources. At the same time, science outside EA could also benefit from having relevant effects knowledge tested and refined through site-specific application inside EA. In sum, both scientific communities, through ongoing collaboration and experimentation, could mutually support one another in the common pursuit of reliable knowledge and sustainable development decisions.

7.4 Impact Prediction

7.4.1 Prediction and Uncertainty in EA

For my purposes, the terms forecast, projection, and prediction are treated as synonyms. Each connotes a statement specifying the expected future condition of a thing of interest. In EA, the things of interest are specified components of the environment, including both objects (e.g., wildlife, soil nutrient pools) and processes (e.g., predation, nutrient cycling). Predictions are always conditional statements, and are stated in terms of the units of measurement used to define the starting condition of the thing of interest. They take the following general form: if specific relationships and starting conditions hold, then the thing of interest is expected to have the following conditions through a specified future time at specified locations. In EA, the relationships and starting conditions must include the undertaking being assessed.

The earliest frameworks for EA (e.g., Leopold et al. 1971; Dee et al. 1973) treated the notion of impact prediction trivially, relying on pseudo-quantitative ranking and scoring methods to convey the intuition of analysts and experts. Subsequent commentaries and guidance materials for EA (e.g., Andrews 1973; Carpenter 1976; Schindler 1976; Holling 1978; Ward 1978; Munn 1979) criticized earlier methods, calling for more rigorous approaches to impact prediction based on ecological knowledge, experimentation, and simulation modelling. Recognizing the fundamental uncertainty surrounding complex ecosystems, Holling (1978) maintained that it would be impossible to predict the environmental impacts of development with absolute certainty. He argued, therefore, that the purpose of impact prediction should be to explicitly identify and reduce key uncertainties during the early stages of development planning and design. In this way, critical uncertainties might have a better chance of being effectively managed and reduced through ongoing monitoring and follow-up.

In addition to experimentation, Holling (1978) outlined a number of techniques that might be used to test predictions and characterize lingering uncertainties prior to decision-making. First, he outlined a process of "invalidation", in which a model's predictive outputs are compared with the effects of past perturbations in similar ecosystems, thereby testing and even expanding the limits of a model's credibility. Second, he described a process of sensitivity analysis, in which a model's fixed parameters are adjusted individually and then simultaneously in order to test the responsiveness of predictions to parameter uncertainty. Lastly, he described a process of generating alternative models, in which analysts explore the consequences of adding or removing potentially influential (yet extremely uncertain) processes or driving variables. According to Holling (1978), the end result would be a small set of plausible models that consider all potentially relevant factors, provide reliable impact predictions, yet explicitly preserve remaining uncertainties for ongoing management and reduction.

During the 1980s, a number of commentaries and reviews (e.g., Rosenberg et al. 1981; Caldwell et al. 1982; Beanlands and Duinker 1983; Clark et al. 1983; Culhane 1987) continued to highlight major shortcomings in formal EA practice, particularly with respect to impact prediction. In general, predictions were found to be vague, untestable, or otherwise non-existent. In a Canadian context, Beanlands and Duinker (1983) outlined the following general standards for impact prediction in EA: (i) predictions should be stated explicitly in testable, quantitative terms, (ii) the timing and spatial extent of predicted impacts should be specified, (iii) the range of uncertainty in impact predictions should be specified, and (iv) the conceptual and technical basis for impact predictions should be made clear.

Duinker and Baskerville (1986) later expanded on these general standards by outlining a somewhat more detailed framework for impact prediction in EA. Amongst other things, Duinker and Baskerville (1986) highlighted the following as core elements of a rigorous, scientific approach: (i) quantitative, process-based, feedback-type models, (ii) explicit connections between development actions and environmental components of interest, (iii) a minimum of two predictions (one with the proposed development in place and one without), (iv) an appropriate level of spatial and temporal resolution, and (v) use of sensitivity analyses to characterize remaining uncertainties. Together with earlier guidance materials, the principles and protocols outlined by Beanlands and Duinker (1983) and Duinker and Baskerville (1986) established a strong basis for making useful and defensible impact predictions in EA.

While the topic of environmental impact prediction has continued to receive attention in the literature of the 1990s and 2000s (e.g., Armstrong 1999; Clark et al. 2001; Demyanov et al. 2006), the basic standards for prediction outlined in the literature of the 1980s have not been challenged or expanded. Cashmore (2004), however, has recently questioned the role of science in EA, arguing for a re-conception of EA as a largely political process. In response, Greig and Duinker (2011) have argued for an ongoing pivotal role for science in EA, that is, to provide useful and defensible predictions of environmental impact.

In a similar vein, the issue of uncertainty (and how best to deal with it) has been a topic of considerable discussion in the recent EA literature, particularly with respect to modelling and impact prediction. Reviews of theory (e.g., Rotmans et al. 2001; Walker et al. 2003; Refsgaard et al. 2007; van der Sluijs 2007; Warmink et al. 2010; Bond et al. 2015; Gregr and Chan 2015; Leung et al. 2015; Cardenas and Halman 2016) and practice (e.g., Tennøy et al. 2006; Duncan 2008; Wardekker et al. 2008; Petersen et al. 2011; Larsen et al. 2013; Leung et al. 2016; Lees et al. 2016) have together highlighted major shortcomings in the characterization, disclosure, and ongoing reduction of scientific uncertainty in EA.

To improve explorations of the future, Duinker and Greig (2007) recently proposed greater use of scenarios and scenario analysis in EA. In general, they argue that scenario analysis provides a crucial means of grappling with uncertainties inherent in the prediction of environmental impacts. In other words, to broaden the scope and horizon of predictive efforts in EA, scenario analysis might be used to explore the consequences of potentially influential (yet highly uncertain) driving variables, such as climate change, induced development, and changing market demands. According to Duinker and Greig (2007), the minimum of two predictions needed to define an environmental impact (i.e., one with the proposed development in place and one without) should be expanded to include a range of plausible future scenarios.

Overall, the prediction of environmental impacts, along with the explicit communication and reduction of scientific uncertainty, continue to be recognized as core elements of EA's contribution to sustainable development. In the following section I review the application of science to environmental impact prediction, as described in the peer-reviewed scientific literature published since the 1970s. Although my review of the literature focuses on the science of environmental impact prediction outside formal EA, some examples of regulatory practice have inevitably found their way into the literature reviewed here.

7.4.2 Using Science to Predict Environmental Impacts

In the 1970s, a wide range of hydrological models were used to predict the effects of point-source pollution on water quality. Examples included a model for predicting the effects of highway de-icing salts on groundwater quality (Gelhar and Wilson 1974), a model for predicting the effects of mine drainage on stream water quality (Herricks et al. 1975), and a model for predicting the effects of sewage discharges on lake water quality (Canale et al. 1973). Likewise, a range of atmospheric dispersion models were used to predict the effects of point-source pollution on air quality. Examples included a model for predicting the effects of airport traffic on local air quality (Daniels and Bach 1976), and a model for predicting the downwind effects of thermal power plant emissions (Carpenter et al. 1971). Indeed, dispersion models were also used to predict the physical transport and fate of marine oil spills (Blaikley et al. 1977).

In addition to predicting changes in air and water, a number of models were used to predict the effects of industrial food and timber production on soils. Examples included a model for predicting the effects of agricultural irrigation on soil nitrogen and phosphorus pools (Dutt et al. 1972), and a model for predicting the effects of timber production on forest floor organic matter and nitrogen pools (Aber et al. 1978). Finally, the 1970s saw the use of simulation models to predict the effects of pollution, harvesting, and physical infrastructure on fish and wildlife populations. Examples included a model for predicting the effects of power plant cooling systems on freshwater fish populations (DeAngelis et al. 1977), a model for predicting the effects of exploitation and enhancement on wild salmon populations (Peterman 1975), and a model for predicting the effects of pollution, harvesting, and hydroelectric development on fish and wildlife populations (Walters 1974).

By the 1980s, EA scholars and practitioners (e.g., Beanlands et al. 1985; Peterson et al. 1987; Sonntag et al. 1987) had begun to emphasize the importance of predicting cumulative environmental effects, that is, the effects of multiple natural and anthropogenic stressors acting simultaneously. Indeed, a number of simulation models were used to explore the environmental consequences of multiple development activities. Examples included a model for predicting the effects of harvesting, stocking, and lamprey reduction on trout populations (Walters et al. 1980), a model for predicting the effects of fire and wildlife management on tree canopy regeneration (Pellew 1983), a model for predicting the effects of exploitation and water pollution on freshwater fish populations (Goodyear 1985), and a model for predicting the effects of low-flying aircraft and coyote predation on a threatened pelican population (Brunnell et al. 1981).

In addition to more complex models, simpler models were used to predict the individual environmental effects of particular development actions. Examples included models for predicting the effects of oil spills on marine fish and mammal populations (Ford et al. 1982; Spaulding et al. 1983; Reed et al. 1984; Reed et al. 1989), models for predicting the effects of power plant cooling systems on marine and freshwater fish populations (Jensen 1982; Shuter et al. 1985; Polgar et al. 1988), and models for predicting the effects of exploitation on the growth and productivity of forests, fish, and wildlife resources (Deriso 1980; Shugart et al. 1980; Williams 1981; Vanclay 1989).

The 1980s also saw the first use of spatially explicit simulation models for predicting the effects of human developments on terrestrial landscapes. Examples included a model for predicting the effects of hydroelectric development on water flows and vegetation succession in a forested floodplain (Pearlstine et al. 1985), and a model for predicting the effects of canals, levees, and sea level rise on water and sediment transport in a coastal wetland (Sklar et al. 1985).

In the 1990s and 2000s, the science of environmental impact prediction expanded greatly, with a growing emphasis on the broad-scale consequences of land-use change, habitat fragmentation, and global climate change. For instance, dynamic landscape models have been used to predict the effects of industrial timber harvesting and climate change on the long-term productivity and composition of forested watersheds (e.g., Scheller and Mladenoff 2005; Steenberg et al. 2011). Such models have also been linked with wildlife population models to predict the effects of timber harvesting and climate change on the dispersal and survival of threatened animal populations (e.g., Larson et al. 2004; Akçakaya et al. 2005; McRae et al. 2008). In terms of biodiversity, forest landscape models have been used to predict the effects of timber production and other management practices on remaining habitat structure and suitability (e.g., Marzluff et al. 2002; Shifley et al. 2006; Radeloff et al. 2006; Gustafson et al. 2007). In a similar vein, a variety of carbon budget models have been used to predict the effects of timber production, climate change, and insect outbreaks on terrestrial carbon stocks (e.g., Karjalainen et al. 2002; Seidl et al. 2008; Kurz et al. 2008).

At the strategic level, various hydrological and soil process models have been used to predict the effects of land-use and climate change on the water and nutrient budgets of watersheds (e.g., Ferrier et al. 1995; Johnes 1996; Niehoff et al. 2002; Mango et al. 2011). Likewise, a variety of hydrological and microclimatic models have been used to predict the effects urban development on local water and energy budgets (e.g., Jia et al. 2002; Rosenzweig et al. 2009; Velazquez-Lozada et al. 2006). For marine environments, coupled ocean-biogeochemical-population models have been used to predict the effects of exploitation and climate change on the productivity of fish stocks (e.g., Lehodey et al. 2013; Lindegren et al. 2010). Similar models have also been used to predict the effects of fish harvesting and climate change on the productivity of coupled fish-coral ecosystems (e.g., Mclanahan 1995; Mumby 2006; Sebastián and McClanahan 2012; Weijerman et al. 2015).

In terms of new and emerging industries, a variety of simulation models have been used to predict the effects of fish farms on marine and freshwater quality (e.g., Håkanson and Carlsson 1998; Wild-Allen et al. 2010), the effects of shale gas development on groundwater quality (e.g., Schwartz 2015; Reagan et al. 2015), and the effects of wind energy developments on migratory bird populations (e.g., Eichhorn et al. 2012; Bastos et al. 2015). Still, an immense variety of simulation models have been used to predict the environmental impacts of more traditional developments. Examples include models for predicting the effects of roads and other noisy infrastructure on wildlife populations (e.g., Weclaw and Hudson 2004), the effects of mining and mineral processing on air and water quality (e.g., Herr et al. 2003; Dutta et al. 2004), and the effects of dams and artificial reservoir management on fish populations (e.g., Jager et al. 1997; McCleave 2001).

Again, as pointed out by Greig and Duinker (2011), science outside EA has developed the capacity to generate powerful insights into ecological processes and cause-effect relationships. Indeed, the scientific literature describes the development and application of an abundance of process-based models for environmental impact prediction. In my view, the scientific enterprise situated inside EA could benefit substantially by calling in such knowledge. Likewise, the scientific enterprise situated outside EA could benefit from having such knowledge applied and refined through site-specific application inside EA.

7.5 Impact Significance Determination

7.5.1 Overview

In EA, 'significance' refers to the overall acceptability, legality, or sustainability of predicted environmental impacts, typically in terms of societal norms, regulatory standards, ecological thresholds, or some combination of the three (e.g., Lawrence 2007a, 2007b, 2007c). Ultimately, the significance attributed to predicted impacts will play a key role in determining whether a proposed development will receive final regulatory approval. The determination of impact significance is therefore closely linked to the evaluation of development alternatives, formal reviews, and regulatory decision-making. While the bulk of scientific work in EA is directed at generating reliable impact predictions, the determination of impact significance represents the critical stage at which the acceptability of future VEC conditions is judged. As such, significance evaluation is arguably one of the more challenging and disputed aspects of an EA. According to Lawrence (2007a), there are two general approaches to impact significance determination: technical and collaborative. A third approach, called reasoned argumentation, seeks to integrate technical and collaborative approaches in order to achieve more-substantiated conclusions that all stakeholders can contribute to.

Here I review the formal body of literature on impact significance in EA, with an emphasis on both technical and collaborative approaches to significance determination.

7.5.2 Integrating Technical and Collaborative Approaches

The earliest frameworks for EA (e.g., Leopold et al. 1971; Dee et al. 1973) relied on numerical weighting and scoring methods to assign 'importance' values to potential development-environment interactions. Subsequent guidance materials for EA (e.g., Sharma et al. 1976) sought to provide more rigorous definitions of the term 'significance', and to establish basic criteria (e.g., magnitude, extent, duration, frequency) for determining impact significance in EA practice (e.g., Andrews et al. 1977). At the same time, members of the scientific community (e.g., Cairns 1977; Holling 1978) advocated concepts like 'assimilative capacity' and 'stability domains' when trying to gauge the ability of ecosystems to absorb or recover from anthropogenic disturbances. Still other authors (e.g., Bissett 1978) criticised early EA frameworks for deliberately excluding diverse stakeholder perspectives from the evaluation of impact 'importance' and, ultimately, from influencing development decisions.

Reviews of significance determination literature in the 1980s (e.g., Beanlands and Duinker 1983; Duinker and Beanlands 1986) outlined a range of technical, political, and regulatory perspectives on the subject. In general terms, Duinker and Beanlands (1986) highlighted some differences between significance judgements based on ecological limitations, limits of social acceptability, and regulatory environmental standards. According to Duinker and Beanlands (1986), a rigorous approach to significance determination should consider at minimum: (i) the social importance of the environmental attribute in question, (ii) the magnitude and distribution of predicted environmental impacts, and (iii) the reliability of impact predictions. From a scientific perspective, Conover et al. (1985) proposed a framework for evaluating impact significance based on the application of context-specific, ecological thresholds. At the same time, other authors proposed frameworks based on the application of more socially-derived 'thresholds of concern' (e.g., Sassaman 1981; Haug et al. 1984).

More recently, the determination of impact significance has continued to be an important topic of discussion in the literature, with authors advocating a variety of tools and techniques. Examples include the use of so-called ‘decision-trees’ (e.g., Sippe 1999), ‘fuzzy sets’ (e.g., Silvert 1997; Wood et al. 2007), matrices (e.g., Ijäs et al. 2010; Toro 2012), multi-criteria decision methods (e.g., Cloquell-Ballester et al. 2007), spatial indices (e.g., Antunes et al. 2001), regulatory environmental standards (e.g., Kjellerup 1999; Schmidt et al. 2008), socially-derived thresholds of acceptability (e.g., Canter and Canty 1993; Ehrlich and Ross 2015), and ecologically-based response thresholds (e.g., Duinker and Greig 2006; Duinker et al. 2013). Though the purpose of significance determination has largely been recaptured in terms of sustainable development (e.g., Gibson 2001), generic criteria such as ‘magnitude’, ‘extent’, ‘frequency, and ‘duration’ have seen continued use throughout the literature. Despite this growing body of literature on the subject, reviews of theory (e.g., Thompson 1990; Lawrence 2007b, 2007c; Jones and Morrison-Saunders 2016) and practice (e.g., Sadler 1996; Barnes et al. 2002; Wood 2008; Khadka et al. 2011; Briggs and Hudson 2013) have highlighted ongoing shortcomings in the evaluation of impact significance.

To reconcile technical and collaborative contributions to significance determination, Lawrence (2007a) has proposed a more integrated approach. While the technical approach aims to judge impact significance quantitatively based on scientific analysis and knowledge, the collaborative approach aims to do so qualitatively based on community knowledge and perspectives. A third approach, called reasoned argumentation, seeks to integrate quantitative and qualitative approaches. According to Lawrence (2007a), “The overall intent is to contribute to significance determination procedures that are less biased and distorted, more fully substantiated, more open, inclusive and collaborative and more effectively linked to decision making”. In my view, the reasoned argumentation approach represents a much stronger framework for significance determination than either the technical or the collaborative approach used alone.

7.6 Evaluation of Alternatives

7.6.1 Overview

Closely related to the evaluation of impact significance is the evaluation and selection of development alternatives. As pointed out by Wood (1995), decision-making in EA may involve relatively simple yes/no regulatory approvals, or more complex decisions involving a choice among alternatives. According to Lawrence (1993), when development decisions involve multiple alternatives, environmental effects, and stakeholder perspectives, formal evaluation tools may be needed to handle a large number of trade-offs, but also to ensure transparency in the selection of a preferred alternative. Lawrence (1993) points out that there are two broad approaches to evaluating development alternatives in EA: quantitative and qualitative. Therein, the quantitative approach is to evaluate alternatives numerically, relying primarily on abstract weighting, scaling, and aggregation techniques. The qualitative approach aims to evaluate development alternatives deliberatively, incorporating a range of stakeholder values and perspectives. Similar to the 'reasoned argumentation' approach to impact significance determination (Lawrence 2007a), Lawrence (1993) proposed the integration of quantitative and qualitative approaches to the evaluation and selection of development alternatives. Such an approach, he argues, aims to contribute to more-substantiated and democratic environmental decisions.

In this section I review the literature on evaluating development alternatives in EA, with coverage of both quantitative and qualitative approaches.

7.6.2 Integrating Quantitative and Qualitative Approaches

The earliest frameworks for EA served as rapid evaluation tools for numerically weighting, aggregating, and comparing potential development-environment interactions. While early checklist/matrix approaches (e.g., Leopold et al. 1971; Dee et al. 1973) relied on the use of dimensionless impact scores to quantify development-environment interactions, cost-benefit analysis (e.g., Kasper 1977; Bohm and Henry 1979) required all environmental impacts to be translated into monetary terms, aggregated, and subtracted from benefits. Both cost-benefit analysis and early matrix/checklist methods received much criticism in the early literature (e.g., Lapping 1975; Pearce 1976; Baram 1980). From both technical and regulatory perspectives, Andrews (1973) criticised such approaches for relying on the use of arbitrary, dimensionless impact scores or dollar amounts to quantify trade-offs. From a political perspective, Bisset (1978) criticised such approaches for being undemocratic and for concealing the basis upon which numerical values are assigned to impacts by experts.

Early scientific guidance materials for EA (e.g., Holling 1978; Munn 1979) proposed somewhat more rigorous approaches to evaluating development alternatives based on the predicted future condition of actual measurable environmental attributes (i.e., indicators). Holling (1978) observed that when the number of development alternatives and indicators is large, it may be helpful to quantify the trade-off preferences of decision-makers and use techniques of mathematical optimization to determine the 'best' possible alternative. He also noted, however, that such complicated evaluation techniques may often be unnecessary or even counterproductive. Holling (1978) concluded that although quantitative evaluation techniques may be helpful when decision-makers are faced with a large number of alternative developments and indicators, a simple visual examination of a side-by-side time-series or other graphical depiction of indicator patterns should generally suffice. According to Holling (1978), such a qualitative approach to evaluation helps to ensure that all parties are able to judge the predicted impacts of development alternatives from their own perspectives.

In the 1980s, the participatory dimension of EA began to receive more critical attention in the literature (e.g., Rosenberg et al. 1981; Rossini and Porter 1982). These authors called for more-inclusive approaches to conducting EAs and to making environmental decisions. In place of more-traditional approaches to making group decisions and settling disputes (i.e., quasi-judicial hearings, litigation), a number of consensus-building techniques were proposed for the evaluation and selection of development alternatives. Typical examples included the use of mediation (e.g., Cormick 1980; Watson and Danielson 1983; Sorensen et al. 1984) and the use of Delphi method (e.g., Lowe and Lewis 1981; Richey et al. 1985; Miller and Cuff 1986). Still, with the rapid expansion of more powerful computers, a number of authors continued to advocate the use of quantitative ‘aids’ for structuring complex decisions. Examples included so-called ‘expert’ or ‘decision support’ systems (e.g., Hushon 1987; Lein 1989), multi-criteria decision methods (e.g., Hobbs 1980; Hobbs 1985; Nijkamp 1986), and optimal control theory (e.g., Walters 1986).

At the same time, a number of authors continued to highlight the inadequacies of both quantitative and qualitative approaches to evaluating development alternatives (e.g., McAllister 1980; Hollick 1981; Bakus et al. 1982). To move towards a more-inclusive, transparent, and less-biased approach to environmental decision-making, authors like Bakus et al. (1982) and Edwards and von Winterfeldt (1987) proposed frameworks that would use multiple-criteria decision methods as a basis for structuring stakeholder consensus-building and negotiation processes

In many ways, the literature of the 1990s and 2000s has been a continuation of the earlier literature on evaluation and decision-making in EA. With respect to quantitative approaches, there has been an increasing focus on multi-criteria decision methods for evaluating alternatives (e.g., Lahdelma et al. 2000; Janssen 2001; Huth et al. 2004; Pohekar and Ramachandran 2004; Hyde et al. 2005; Kain and Söderberg 2008). At the same time, there has been continued criticism of traditional quantitative techniques like cost-benefit analysis (e.g., Munda 1996; Brunner and Starkl 2004).

With respect to qualitative approaches, a number of authors (e.g., Lee 1993; Gibson 2006) have continued to highlight the use of rule-based negotiation and mediation, while other authors (e.g., Kingston et al. 2000; Konisky and Beierle 2001) have proposed more innovative arrangements like citizen juries, study circles, roundtables, and web-based forums. Still, reviews of decision-making theory (e.g., Ortolano and Shepherd 1995; Kørnøv and Thissen 2000; Nilsson and Dalkmann 2001; Gibson 2006; Pischke and Cashmore 2006) and practice (e.g., Sadler 1996; van Breda and Dijkema 1998; Glasson 1999; Morrison-Saunders and Bailey 2000; Leknes 2001; Sager 2001; Richardson 2005) have highlighted ongoing weaknesses in the link between EA and decision-making.

To enhance the influence of EA on development planning and decision-making, many authors (e.g., Sinclair and Diduck 1995; Shepherd and Bowler 1997; Petts 1999;

Doelle and Sinclair 2006; Stewart and Sinclair 2007; Dietz and Stern 2008) have called for stronger participatory processes that encourage ‘early and ongoing’ stakeholder participation, particularly with respect to the design, evaluation, and selection of development alternatives.

In order to reconcile differing perspectives on the subject, Lawrence (1993) proposed the integration of qualitative and quantitative approaches to evaluation and decision-making in EA. Such an integrated approach, he argues, would provide more substantiated and democratic conclusions than either of the two approaches used alone. Indeed, a number of authors have proposed the use of multi-criteria decision analysis to support more structured and deliberative group decision-making processes (e.g., Ramanathan 2001; Petts 2003; Mardle et al. 2004; Bojórquez-Tapia et al. 2005; Dietz and Stern 2008; Kalbar et al. 2013). In my view, combining such quantitative and qualitative approaches to evaluation may be helpful for achieving mutually-agreeable solutions when decision-makers are faced with large numbers of development alternatives, environmental impacts, and stakeholder objectives.

7.7 Formal Reviews

7.7.1 Overview

Founded on principles of public scrutiny, EA processes generally allow for public review of EA documents prior to regulatory decision-making. In many jurisdictions, environmental laws and regulations require EAs for large and controversial undertakings to be reviewed by independent panels (e.g., Wood 1995; Sadler 1996; Ross 2000). Such panels typically evaluate an EA’s overall quality against established ‘scoping guidelines’ or ‘terms of reference’, and then submit a set of recommendations to regulatory decision-makers. To inform such recommendations, review panels typically hold formal review hearings centred on the evaluation of a draft EA report. Here public comments are invited, expert testimony is received, and additional information is requested from proponents. At the end of the review process, the proponent publishes a final EA report and the review panel submits its final recommendations to regulatory decision-makers.

In this section I briefly summarize the literature on formal EA document reviews, focusing on the intersection of administrative, scientific, and political dimensions.

7.7.2 Administrative, Scientific, and Political Dimensions

The first EA processes in North America relied primarily on administrative and judicial mechanisms for review (e.g., Leventhal 1974). According to the literature (e.g., Lynch 1972; Jacobsen 1978; Karp 1978), such reviews were often the source of much confusion and frustration among EA participants due to a lack of general standards for judging the ‘adequacy’ of draft EAs. This typically resulted in prolonged and costly disagreement among EA participants. Indeed, the sprawling and descriptive nature of early EA reports was often attributed to this threat of judicial review (e.g., Wichelman 1976). Generally, such arguments postulated that proponents might intentionally prepare encyclopedic EAs, thereby giving the impression of comprehensive coverage while simultaneously obscuring relevant or controversial environmental impacts. Indeed, the prevailing yet unrealistic expectation of EA administrators and reviewers for ‘complete’ environmental information was also recognized in the literature (e.g., Andrews 1973; Carpenter 1976).

To encourage early agreement among all EA participants on the design, preparation, and review of assessments, EA administrators in North America introduced the first formal requirements for ‘scoping’ in the late 1970s (e.g., Fisher 1979; Hourcle 1979). In general, such procedures were intended to foster a more focused and collaborative approach to designing, conducting, and inevitably reviewing the quality of EAs.

In the early 1980s, some authors (e.g., Beanlands and Duinker 1983; Carpenter 1983) lamented the costly, reactive, and generally frustrating nature of formal EA reviews. Moreover, both Carpenter (1983) and Beanlands and Duinker (1983) observed that the sprawling character of formal scoping guidelines had seemed to perpetuate many of the problems formerly attributed to adversarial review hearings. The lingering issue, they argued, was the failure of EA practitioners and members of the scientific community to come together and agree on a common basis for designing, conducting, and reviewing scientific contributions to EA. In other words, they observed that there was still no widespread agreement—even amongst scientists themselves—as to what could reasonably be accomplished by science in EA. What they could agree on, however, was the need for greater focus and the need for basic scientific standards. In the Canadian context, Beanlands and Duinker (1983) outlined six so-called ‘requirements’ that were to serve as basic principles for both the design and evaluation of scientific contributions to EA.

Later, Ross (1987) outlined the roles of different evaluators in judging the three major aspects of an EA report: (i) focus, (ii) scientific and technical soundness, and (iii) clarity. According to Ross (1987), it should be the role of scientific and technical experts to judge matters of science, the role of the public to evaluate matters of focus and clarity, the role of administrators to test EAs against scoping guidelines, and the role of independent panels to synthesize all comments and contributions into a set of recommendations. In sum, an ideal EA should be focused on stakeholder-relevant issues, scientifically sound, and yet readily understandable to technical specialists and lay stakeholders alike.

According to Wood (1995), “If there is only one point in an EIA process where formal consultation and participation take place it is during the review of the EIA report”. He goes to explain how in many jurisdictions, ‘public review’ is virtually synonymous with public participation. According to a number of authors (e.g., Diduck and Sinclair 2002; Petts 2003; Doelle and Sinclair 2006; Stewart and Sinclair 2007), such formal mechanisms have generally failed to provide citizens with an opportunity to influence development decisions, and have thus failed to satisfy basic principles of participatory democracy. Moreover, they argue that formal procedures for citizen engagement may even represent barriers to participatory environmental decision-making. Shepherd and Bowler (1997) have argued that in order to improve the quality of participatory practices in EA, proponents must go beyond formal requirements simply to consult the public before and after an EA is prepared. Furthermore, they argue that it is often proponents who stand to benefit the most from inviting all stakeholders to participate in the design, evaluation, and selection of a preferred development alternative.

Still, the formal review of draft EAs as a means of quality control continues to be discussed in the literature (e.g., Lee and Brown 1992; Ross 2000; Ross et al. 2006). More recently, the so-called ‘review package’ outlined by Lee and Colley (1991) has become a popular method for reviewing and rating the quality of EA documents (e.g., Cashmore et al. 2002; Canelas et al. 2005). Põder and Lukki (2011), however, have criticised such methods for emphasizing ‘completeness’ of information over actual information quality. According to Ross et al. (2006), “The principle for preparing an EIS is simple; it should present a clear, concise summary of the likely environmental impacts, the proposed mitigation measures, the significance of the residual impacts and suggestions for needed follow-up studies”.

7.8 Follow-Up

7.8.1 Overview

According to Morrison-Saunders and Arts (2004), “A key feature of EIA is that its deals with the future and consequently is intrinsically uncertain. Follow-up addresses the uncertainties in EIA”. Arts et al. (2001) identify four major activities within EA follow-up: (i) monitoring, (ii) evaluation, (iii) management, and (iv) communication. Therein, monitoring is the ongoing collection of data, evaluation is the appraisal of those data’s conformance with expectations, management is the set of actions taken in response to those evaluations, and communication is the disclosure of results to stakeholders. Arts et al. (2001) outline five major objectives for EA follow-up: (i) provide information about the consequences of an activity, (ii) enhance scientific knowledge about environmental systems and cause-effect relationships, (iii) improve the quality of assessment methods and techniques, (iv) improve public awareness about the effects of development, and (v) maintain decision-making flexibility.

In this section I provide two short reviews: one of the general literature on EA follow-up and another focusing more specifically on the scientific literature surrounding environmental effects monitoring.

7.8.2 Effects Monitoring, Adaptive Management, and Participation

Holling (1978) argued that although accurate prediction of environmental impacts may be fundamentally impossible, some “postdiction” (i.e., monitoring) might contribute to better predictions in the future. Holling (1978) further argued that effects monitoring would provide an important opportunity to pursue model “invalidation”, thereby reducing key uncertainties in the long-run. To ensure the responsiveness of developments to new ecological information, Holling (1978) proposed an integrated brand of management and science called “adaptive management”. In Holling’s (1978) adaptive management framework, not only is modelling and impact prediction integrated with development planning and design, but also with long-term ecological monitoring and management. According to Holling (1978), “Adaptive management can take a more active form by using the project itself as an experimental probe”. He went on to write “An explicit attempt to use the project itself can be used to address one element of the uncertainty surrounding environmental responses”. Holling (1978) was also one of the first to note the difference between regulatory monitoring, aimed at ensuring compliance with development approval conditions (e.g., mitigation measures), and scientific monitoring, aimed at testing impact predictions to reduce key uncertainties.

One of the six ‘requirements’ outlined by Beanlands and Duinker (1983) was to monitor the effects of developments on VECs. Such monitoring efforts would be aimed at testing impact predictions, thus improving ecological effects knowledge for future EAs. One aspect of effects monitoring to receive particularly close attention in the 1980s was the statistical design of field sampling programs, as well as the challenges of separating development-induced changes from natural variation. Whereas some authors (e.g., Green 1989) advocated simple ‘before-after’ or ‘control-impact’ comparisons to define an environmental impact, other authors (e.g., Stewart-Oaten et al. 1986) advocated a so-called ‘before-after-control-impact’ (BACI) design. According to Stewart-Oaten et al.’s (1986) BACI framework, an environmental impact is defined as a change in the difference between measured environmental conditions at a control site and a treatment site following development. Duinker (1989) later argued that the establishment of such spatial and temporal ‘controls’ is often impossible, particularly when dealing with highly mobile animal populations. Duinker (1989) reasoned that an environmental impact can only realistically be defined as the difference between the predicted future condition of an environmental component of interest with and without the proposed development in place. Consequently, Duinker (1989) concluded that monitoring can only realistically replace one of the predicted future time-series in a "difference calculation of impact".

The participatory dimension of EA follow-up has recently been a topic of considerable discussion in the literature (e.g., Hunsberger et al. 2005; Morrison-Saunders and Arts 2005). In general, it is argued that enhancing stakeholder participation in EA follow-up increases the utility of an assessment to all parties involved. According to Morrison-Saunders and Arts (2004), "Some follow-up programmes extend beyond simple communication to specifically include direct stakeholder participation in the monitoring, evaluation, and management steps as well". Indeed, the literature describes a range of community-based (e.g., Hunsberger et al. 2005; Lawe et al. 2005; O'Faircheallaigh 2007) and proponent-led (e.g., Marshall 2002; Marshall 2005; Noble and Storey 2005; Noble and Birk 2011; Devlin and Tubino 2012) EA follow-up programs. According to advocates of a community-led approach, such arrangements offer a level of transparency and trust not possible under proponent-driven initiatives. Most authors now agree that involving all stakeholders in follow-up processes allows EA to have a greater influence on development outcomes as well as the long-term adaptive management of environmental effects. Indeed, established principles for good follow-up (e.g., Arts et al. 2001; Morrison-Saunders and Arts 2004; Marshall et al. 2005) have emphasized regulatory, scientific, and participatory dimensions of EA.

From a scientific perspective, a number of papers on environmental effects monitoring (e.g., Adams 2003; Hewitt et al. 2003) have recently focused on the need to explicitly link measured environmental effects with particular causes (i.e., stressors), thus reducing uncertainty in cause-effect relationships. It is generally argued that such approaches to causal inference are most useful when dealing with numerous, confounding, or otherwise uncertain causal variables (e.g., complex mill effluents). Indeed, a number of authors (e.g., Roux 1999; Dubé and Munkittrick 2001; Kilgour et al. 2007) have recently proposed frameworks that integrate so-called 'stressor-based' and 'response-based' approaches to environmental effects monitoring. According to Roux

(1999), the stressor-based approach focuses on setting and enforcing regulatory standards for controlling levels or concentrations of particular stressors. Conversely, the response-based approach is focused on measuring ecological conditions in the presence of development. According to Roux (1999), integrated stressor-response monitoring simultaneously measures both stressor and effect variables during development, thus providing scientific evidence to inform the adaptive management of stressors.

A number of other authors (e.g., Lowell et al. 2000; Adams 2003) have recently outlined protocols for a so-called 'weight-of-evidence' approach to inferring causality in environmental effects monitoring. Adams (2003) outlines seven 'causal criteria' for evaluating a potential relationship between a stressor and an observed effect: (i) strength of association between stressor and effect variables, (ii) consistency of association between stressor and effect variables, (iii) specificity of association between stressor and effect variables, (iv) time order of stressor and effect measurement, (v) spatial or temporal dose-response gradient, (vi) experimental evidence, and (vii) a scientifically plausible explanation for a mechanism linking the proposed cause and effect.

According to Greig and Duinker (2011), it is the role of science inside EA to predict and monitor the environmental effects of developments, thereby testing and refining the predictive tools developed by science outside EA. In this way, environmental effects monitoring provides the critical feedback needed by science outside EA to contribute better cause-effect knowledge in the future. Though the scientific literature reveals a growing body of knowledge surrounding the environmental effects of human developments, several authors (e.g., Karkkainen 2008; Bjorkland, 2013; Roach and Walker 2017) observe ongoing inadequacies in how environmental effects monitoring is practiced in EA. In this last section, I provide a general outline of the environmental effects knowledge published in the formal scientific literature. While my review highlights a number of studies that conceptually resemble environmental effects monitoring in EA, a few examples from formal EA practice have also been included.

7.8.3 Environmental Effects Knowledge

Most empirical studies of development-induced environmental effects conducted in the 1970s were in the form of brief comparisons of conditions measured at development and 'control' sites. Less common were the results of actual long-term monitoring programs. Examples included studies on the effects of mining and wastewater discharge on marine and freshwater quality (e.g., Balch et al. 1976; Kaufmann et al. 1976; Minear and Tschantz 1976), the effects of hydroelectric and thermal energy generation on freshwater fish (e.g., Merriman and Thorpe 1976; Mathur et al. 1977), the effects of pulp mill effluents on freshwater fish (e.g., Kelso 1977; Leslie and Kelso 1977), the effects of road-kill and hunting on wildlife populations (e.g., McCaffery 1973), and the effects of industrial food and timber production on soil nutrient budgets (e.g., Brown et al. 1973; Olness et al. 1974).

At the strategic level, the 1970s saw the initiation of some of the first monitoring programs aimed at characterizing the broad-scale environmental consequences of persistent air and water pollution. Examples included initiatives for monitoring the accumulation of pollutants like PCBs and methylmercury in freshwater and marine fish (e.g., Munson and Huggett 1972; Walker 1976; Hattula et al. 1978), and programs for monitoring the effects of acid deposition on water, forests, and fish (e.g., Ottar 1976; Galloway et al. 1978).

In the 1980s, empirical research on environmental impacts expanded to include the effects of habitat loss and landscape fragmentation on wildlife. Early examples included studies on the effects of roads, buildings, and other noisy infrastructure on wildlife habitat use (e.g., Witmer and deCalesta 1985; Mattson et al. 1987; McLellan and Shackleton 1988; Vogel 1989; Brody and Pelton 1989; Mech 1989), and the effects of timber harvesting on the abundance and diversity of birds (e.g., Scott and Oldemeyer 1983; Zarnowitz and Manuwal 1985). Still other studies continued to document the effects of hydroelectric and thermal energy developments on fish (e.g., Barnthouse et al. 1983; Bodaly et al. 1984; Madenjian et al. 1986; Muessig et al. 1988), the effects of pulp mill effluents on fish (e.g., Andersson et al. 1988; Sandström and Thoresson 1988), the effects of mining on surface and groundwater quality (e.g., Hill and Price 1983; Coe and

Stowe 1984), and the effects of timber harvesting on watershed nutrient and water budgets (e.g., Sollins and McCorison 1981; Krause 1982; Feller and Kimmins 1984).

The 1980s also saw growing interest in monitoring the environmental effects of fish farming, an emerging industry with uncertain and potentially unwanted environmental consequences. Early examples included studies on the effects of caged fish farming on marine water and sediment quality (e.g., Brown et al. 1987; Kaspar et al. 1988; Eng et al. 1989; Frid et al. 1989).

With the expansion of GIS and satellite remote sensing, the 1980s would eventually see the use of such platforms to monitor the cumulative effects of developments on coastal wetlands (e.g., Nayak et al. 1989). Satellite observation systems would also be used to monitor the effects of deforestation and land-use change on the productivity of tropical forests around the world (e.g., Malingreau et al. 1989).

More recently, there has been an increase in published scientific research on environmental effects. Whereas most empirical effects research published in the 1970s and 1980s was in the form of short-term 'before-after' or 'control-impact' studies, more recent papers (e.g., Roux 1999; Dubé and Munkittrick 2001; Adams 2003, Hewitt et al. 2003; Kilgour 2007) have highlighted the importance of measuring both stressor and response variables at the same time to reduce uncertainty in cause-effect relationships. Noteworthy examples of such an approach include studies on the effects of pulp mill effluent on water and fish (e.g., Culp et al. 2000; Munkittrick et al. 2002; Walker et al. 2002; Hewitt et al. 2005; Dubé et al. 2006; Squires et al. 2010), the effects of mining effluent on water and fish (e.g., Ribey et al. 2002), and the effects of thermal energy developments on fish (e.g., Barnthouse 2000).

The recent literature also highlights a number of studies that have used biotelemetry and satellite remote sensing to characterize the effects of linear developments (e.g., roads) on wildlife-habitat relationships. Examples include studies on the effects of forest roads and cut-blocks on habitat use by wolves (e.g., Whittington et al. 2005; Houle et al. 2009), and studies on the effects of roads, pipelines, and seismic lines on habitat use by caribou (e.g., Dyer et al. 2002; Johnson et al. 2005; Sorensen et al. 2008; Polfus et al. 2011).

Emerging industries and technologies—surrounded by considerable uncertainty with respect to environmental impacts—have also been the subject of empirical scientific research. Examples include studies on the effects of wind energy turbines on birds and bats (e.g., Osborn et al. 2000; Johnson et al. 2003; de Lucas et al. 2004; Masden et al. 2009; Rydell et al. 2010; Plonczkier and Simms 2012), the effects of aquaculture on marine and freshwater quality (e.g., Wu et al. 1994; Boaventura et al. 1997; Selong and Helfrich 1998; La Rosa et al. 2002; Kalantzki and Karakassis 2006), and the effects of shale-gas fracking on surface and groundwater quality (e.g., Fontenot et al. 2013; Olmstead et al. 2013; Warner et al. 2013).

With growing concerns over the potential consequences of global climate change, the recent literature has also highlighted a number of initiatives for monitoring the effects of development on terrestrial carbon stocks. At the project level, such studies may monitor the effects of timber harvesting and forest management on local soil and tree carbon stocks (e.g., Prescott et al. 2000; Finkral and Evans 2008). At the strategic level, such initiatives may monitor the effects of land-use change on national soil and tree carbon stocks (e.g., Scott et al. 2002; Tate et al. 2003; Kurz and Apps 2006; Eckert et al. 2011).

According to Greig and Duinker (2011), "Science inside the EIA process is needed to make specific impact predictions to inform decision-makers of the potential ecological consequences of development alternatives, as well as to measure environmental responses following development start-up for the purpose of model evaluation and refinement". With respect to monitoring, Greig and Duinker (2011) conclude that "Regardless of whether the assignment falls to scientists outside or practitioners inside the EIA process, the monitoring must get done or else reliable knowledge will not, over time, be developed". To this they add, "On the practical side, EIA should be able to experience gradually reduced costs with implementation of strong science in reducing impact uncertainty".

CHAPTER 8 CONCLUSIONS

Previous sections have attempted to address my two broad aims: (i) to provide an overview of scientific developments associated with EA since the early 1970s, as evidenced in the peer-reviewed formal literature, and (ii) to judge, on the basis of evidence found in the literature review, whether scientific theory and practice are at their vanguard in EA-related applications. Here I present a summary of my findings and point to some promising avenues for future research.

Science has been an integral component of EA since its inception in the 1970s. Though this review has uncovered several advancements surrounding EA-related science since that time, it has also confirmed the continued relevance of first- and second-generation guidance materials (e.g., Holling 1978; Beanlands and Duinker 1983). Frameworks for handling scientific uncertainty in EA (e.g., adaptive management, post-normal science) have remained influential in much of the literature that I reviewed, but appear to have had limited application in the context of formal EA practice. Most importantly, perhaps, emerging ecological concepts and imperatives like biodiversity and climate change have begun to expand EA's focus on VECs that are typically selected on the basis of traditional ecosystem elements (e.g., water, air, wildlife). At the same time, related concepts like resilience, thresholds, complexity, and landscape ecology appear to be serving EA studies only in the background. Despite ample evidence of such conceptual and technical developments in the scientific peer-reviewed literature, the general consensus among scholars seems to be that science in EA practice has not kept pace with these advancements.

Attempting to characterize and evaluate scientific advancements in EA practice since the 1970s proved to be somewhat more difficult than my first objective, since most regulatory EA filings do not find their way into the scholarly literature. I did, however, find that most academic reviews have continued to criticize the quality of science practiced in EA since the 1980s (e.g., Fairweather 1994; Treweek 1995; Warnken and Buckley 1998; Greig and Duinker 2011). At the same time, however, many practitioners have reported feeling satisfied with the quality of science in EA, but dissatisfied with the level of importance placed upon it by decision-makers (e.g., Morrison-Saunders and

Bailey 2003; Morrison-Saunders and Sadler 2010). I propose that a more robust inquiry into the quality of science in EA would rely on multiple lines of evidence, including that generated through practitioner surveys, workshops, and EA document reviews.

In addition to highlighting the need for more empirical research around the question of scientific quality in EA, the review has identified several promising avenues for future research in EA-related science:

(i) Delineation of basic cause-effect relationships linking anthropogenic stressors to VECs using both field experiments and monitoring programs (e.g., Johnson et al. 2005; Houle et al. 2009).

(ii) Assembly of cause-effect knowledge into predictive simulation models that can be used to forecast environmental impacts (e.g., Weclaw and Hudson 2004; Kurz et al. 2008).

(iii) Characterization of ecological thresholds associated with the condition of particular VECs (e.g., Richardson et al. 2007; Sorensen et al. 2008).

(iv) Collaborative methods for engaging stakeholders throughout various scientific stages of the EA process (e.g., Mulvihill 2003; Videira et al. 2010; Bond et al. 2015).

(v) Integration of formal scientific knowledge with other forms of knowledge such as traditional and Aboriginal ecological knowledge (e.g., Huntington et al. 2004a, 2004b; Gagnon and Berteaux 2009)

Contemporary debates surrounding science in EA have attributed science-related challenges to several factors. One long-standing argument has been that poor science in EA is the result of inadequate scoping (e.g., Ross et al. 2006; Morrison-Saunders et al. 2014). Others (e.g., Mulvihill 2003; Greig and Duinker 2014) have challenged this notion, pointing out that as long as EA remains a restrictive, proponent-dominated endeavour, scientific contributions and decision outcomes will remain less than satisfactory. Meanwhile, broader debates have centred on finding an appropriate role for science in EA. Whereas some (e.g., Cashmore 2004) have challenged an ongoing role for science in an increasingly politically-driven EA process, others (e.g., Greig and Duinker

2011) have defended it, emphasizing its importance in providing useful and defensible predictions of environmental impact.

Based on this literature review, I am convinced that science remains critical to EA's central task of protecting VEC sustainability, but that science inside EA has not kept pace with developments in science outside EA. I also believe that any improvements to the scientific enterprise inside EA will rely on the adoption of more collaborative, creative, and participatory arrangements for designing and implementing EA-related scientific studies.

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