UNCERTAINTY, LEARNING AND INNOVATION
IN ECOSYSTEM MANAGEMENT

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Modern society faces considerable environmental challenges in the 21st century. Meeting escalating human demands in the face of rising environmental degradation requires substantial reorganization of society's institutional structures. Such transformations typically require fundamental changes in worldviews and values. The emerging science of complex systems is reframing attitudes and approaches to ecosystem management. This dissertation uses a complex systems framework to explore several topical issues.

Chapter 2 examines conflicting scientific information in environmental disputes. Rather than resulting from flawed or fraudulent science, we use heuristic examples to show that conflicting findings may often result from the complex nature of environmental systems and the limitations of analytical tools. Using hierarchical Bayesian techniques we show that pooling multiple studies can enable refined understanding of ecosystems and better estimation of uncertainties.

Chapter 3 asks whether indicators of ecological regime shifts can provide sufficient warning to adapt management to avert regime shifts. Using a fisheries model, we show that large increases in regime shift indicators only occur once a shift is initiated, often too late for aversive management action. To be useful in averting shifts, we conclude that critical indicator levels need to be defined rather than simply detecting change in the indicators.
Chapter 4 uses a social innovation framework to investigate factors that may foster transformation in ecosystem management. Based on analyses of three local-level case studies, we suggest that raising environmental awareness, developing environmental leadership capacity, promoting dialogue between key stakeholders, and providing institutional support may greatly facilitate the emergence and maintenance of integrated, collaborative ecosystem governance institutions.

Chapter 5 provides an example of how “post-normal” scientific approaches have been taught at the graduate level using scenario planning. Key learning outcomes of the course were greater appreciation of the pervasiveness of uncertainty and the need for multiple points of view in understanding complex environmental issues. Most students left the course feeling more positive and inspired about the potential contribution they can make to addressing contemporary environmental challenges.

The final chapter briefly explores cross-cutting themes of uncertainty, integration, and the existence of windows for policy action in ecosystem management and suggests future research directions.
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CHAPTER 1

INTRODUCTION
Introduction

Scientific and technological advances over the past 50 years have dramatically improved human well-being in many parts of the world (MA 2005; UN 2006). However, unanticipated environmental consequences are now emerging that threaten the well-being of current and future generations (MA 2005; Stern 2006). These impacts include, amongst many others, climate change, the health effects of ubiquitous chemical pollutants, and increasing risks of large, abrupt ecological changes (Steffen et al. 2004; MA 2005; IPCC 2007). Such impacts are becoming apparent at a time when there is enormous pressure to expand the benefits of modern society: to millions of people rising out of poverty in emerging economies such as China and India, and the almost 3 billion people (close to half the world’s population) who currently live on less than $2 per day (UN 2006). Adding to these already strenuous demands will be an additional 3 billion people projected to join the global population before it peaks around mid-century (MA 2005; UN 2007).

It is increasingly acknowledged that meeting escalating human demands in the face of rising environmental degradation requires substantial reorganization of modern society (Capra 1984; MA 2005; Stern 2006; IPCC 2007; Loorbach 2007; Martin 2007). However, major social transformations seldom occur without fundamental changes in societal worldviews and values (Kuhn 1970; Argyris & Schön 1978; Hassard 1995). For instance, the transition from the Middle Ages\(^1\) to the Modern, industrial era\(^2\) was largely associated with radical new worldviews and values that emerged during the Enlightenment\(^3\). These new views advocated

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\(^1\) 5\(^{th}\) to 16\(^{th}\) centuries

\(^2\) 18\(^{th}\) century to present

\(^3\) 17\(^{th}\) to 18\(^{th}\) centuries
human reason, rather than divine power, as the primary source of authority in human affairs, resulting in far-reaching changes in basic assumptions and beliefs about the world and humanity's ability to control it (Palmer et al. 2006). Writings from disparate fields suggest that a similar large-scale reframing of worldviews may be currently underway (e.g., Gray 1995; Cortner & Moote 1999; McDonough & Braungart 2002; Waltner-Toews et al. 2008). These changes potentially set the stage for large-scale reorganization of modern society in the 21st century. Particularly influential in reframing how society views and thinks about the management of ecosystems is the emerging science of complex systems (Levin 1998; Miller & Page 2007; Norberg & Cumming 2008; Gros 2008). For the purpose of this dissertation, ecosystem management is seen as the integration of ecological knowledge into broader sociopolitical and values frameworks with the aim of sustaining ecosystems in support of human well-being (Grumbine 1994; Christensen et al. 1996; MA 2005).

A complex systems view

The last three to four decades have seen a revolution in the way scientists think about the world. Ecological, economic, and social systems are no longer viewed as orderly and well-behaved (akin to a machine), but rather as complex and uncertain (more akin to a living organism) (Capra 1997; Cortner & Moote 1999; Kinzig et al. 2006). This shift involves fundamental changes in assumptions about the nature of the world and humanity's ability to understand and manage it. The following characteristics of complex systems are specifically changing societal beliefs about how ecosystems, as complex systems, might best be managed in support of human well-being.
Multiple regimes, non-linearity, and hysteresis. Traditionally, ecosystems were believed to organize around a single equilibrium point ("The balance of Nature"), but both theory and observation have now firmly established that ecosystems, like many complex systems, can exist in several possible regimes (Holling 1973; May 1977; Scheffer et al. 2001; Schröder et al. 2005). Shifts between regimes typically involve large, abrupt, non-linear changes, rather than occurring in smooth, linear ways (Scheffer et al. 2001; Carpenter 2003). Regime shifts in ecosystems are usually precipitated by a combination of a large shock, such as a drought or a flood, and slow changes in underlying variables, often associated with human activities (Beisner et al. 2003; Scheffer and Carpenter 2003). Importantly, the conditions that cause an ecosystem to flip from one regime to another usually differ dramatically from the conditions that cause a shift in the reverse direction, a property known as hysteresis. This characteristic makes many regime shifts difficult or impossible to reverse (Scheffer et al. 2001; Carpenter 2003). Interest in understanding the limits of different regimes has led to the study of ecological resilience, defined as the amount of disturbance that an ecosystem can withstand and still remain in a particular regime (Holling 1973; Carpenter et al. 2001).

The existence of multiple regimes, non-linear responses and hysteresis has important implications for ecosystem management. First, it implies that increasing human demands on the Earth's ecosystems may have large, unanticipated consequences that cannot be predicted based on extrapolation from current experience (MA 2005; IPCC 2007). Second, in addition to the social and economic disruption that regime shifts may cause, the hysteretic nature of many regime shifts means that avoiding unintentional ecological regime shifts is highly desirable (Stern 2006; Norberg & Cumming 2008; Waltner-Toews et al. 2008). These factors
do not justify blind application of the precautionary principle in ecosystem management, but
do point to the need to carefully consider potential risks and costs associated with increased
pressures on ecosystems (Goklany 2001; Harremoes et al. 2002).

*Interactions and emergence.* A distinguishing feature of complex systems is that they are
often defined more by the interactions between their constituent parts than by the parts
themselves. Furthermore, interactions between the components of a complex system give
rise to properties that cannot be predicted from the individual components, a characteristic
known as emergence (Holland 1999; Manson 2001). Emergent properties of ecosystems are
usually evident at multiple scales: for instance, at the level of a cell, a leaf, an individual tree, a
forest, the biome in which it lies, and the Earth (Holling 1992; Gunderson & Holling 2002).
An important implication of emergence is that the properties of a system at a particular scale
cannot be properly understood or managed by focusing on lower-level components or
subsets of the system.

In terms of ecosystem management, interaction and emergence in complex systems have
two important implications. First, the strong interactions that now exist between people and
ecosystems mean that most ecosystems can no longer be adequately understood or managed
as separate from human systems. Rather, to appropriately manage ecosystems in support of
human well-being, the focus needs to be on understanding and governing coupled social-
ecological systems (SES) (Berkes & Folke 1998; MA 2005). Second, the emergent nature of
complex systems highlights the need for integration among different disciplines. SES cannot
be appropriately understood by simply aggregating understanding of the individual subcomponents of the system (Bammer 2005; Norgaard and Baer 2005).

**Self-organization and evolution.** Another important feature of complex systems is that they are self-organizing and constantly evolving. Complex systems, such as ecosystems and SES, consist of a large number of components, with many relationships among them. However, these components and their relationships are not an undifferentiated mass. Relationships within the system are of different strengths, and give internal structure to the system (Holland 1996; Manson 2001). Importantly, this internal structure changes over time, usually in the direction of increasing complexity, and gives rise to a characteristic known as self-organization. Self-organization enables a complex system to evolve over time in ways that enable the system to better interact with its environment (Holland 1996; Camazine et al. 2001). Evolution and change in complex systems does not occur at a constant rate. Rather, short periods of rapid, dramatic change intersperse prolonged periods of relative stability when change is slow and gradual, leading to a phenomenon known as punctuated equilibrium (Eldredge and Gould 1972; Repetto 2006). Varying rates of change can also be understood in terms of the adaptive cycle developed by Holling (1986) and further described by Gunderson and Holling (2002).

Self-organization and evolution have several implications for managing SES. Most fundamentally, it implies that management is focused on a moving target: the system being managed is constantly evolving. In particular, any action taken to manage a SES will influence the trajectory of change in the system. This phenomenon may partly explain why
even the most carefully considered management actions frequently create unanticipated problems after some time. Viewing SES as constantly evolving systems shifts the focus away from searching for "perfect" management policies that can implicitly be implemented indefinitely. Instead, it stresses the need for ongoing learning about the system and ongoing periodic re-evaluation and adaptation of management strategies. This suggests an important change of emphasis in most ecosystem management agencies: rather than trying to maintain ecosystems in a specific state, the focus should be on learning about and managing ecological change along desirable trajectories (Gunderson & Holling 2002; Chapin et al. in press).

**Irreducible uncertainty.** Earlier worldviews held that progressive study could incrementally eliminate uncertainty about the workings of ecosystems. Once ecosystems were thoroughly understood, they could then be finely managed to meet human needs, in much the same way as controlling a complicated machine (Capra 1997; Cortner & Moote 1999). However, it now becoming clear that the complex nature of ecosystems places inherent limitations on our ability to understand, predict, and control ecosystems (e.g., Pilkey & Pilkey-Jarvis 2007; Roe and Baker 2007). For instance, it has been shown that reduction in uncertainties about underlying climate processes cannot further reduce the uncertainty about future temperature increases associated with climate change (Roe and Baker 2007). Furthermore, ongoing evolution in ecosystems and SES continually creates new uncertainties. Understanding of the system gained at one point in time may therefore no longer apply at a future point in time.

Irreducible uncertainty and ongoing evolution over space and time suggest that a fundamental shift in the traditional approach to uncertainty is required. Rather than aiming
to reduce or eliminate uncertainty before taking decisions, uncertainty needs to be explicitly acknowledged and incorporated in environmental decision-making processes. Several new tools have emerged to meet this need. Scenario-planning is an approach that aims to scope possible future trajectories of change in a SES by developing several alternative scenarios of how the future may unfold (van der Heijden 1996; Peterson et al. 2003). As such, it contrasts with more traditional approaches such as forecasting that focus on the accurate prediction of a single future outcome. Large, integrated environmental assessments such as the Millennium Ecosystem Assessment (MA 2005) and the Intergovernmental Panel on Climate Change assessment (IPCC 2007) specifically aim to identify and evaluate the implications of uncertainty to better inform environmental decision-making.

Preview of the chapters

Using a complex systems framework, this dissertation explores several topical issues relating to the management of ecosystems. Following Boyle et al. (2001), ecosystem governance is taken to be the social and political process of defining goals for ecosystem management and resolving trade-offs, while ecosystem management is defined as the actions taken to achieve these goals. The appropriate context for understanding ecosystem management is therefore the scale of coupled SES, and is the scale of analysis adopted in this dissertation.

Chapter 2 examines the issue of conflicting scientific information in environmental disputes and decision-making. Rather than resulting from flawed or fraudulent science (Herrick and Jamieson 2001), we suggest that conflicting findings may often result from the complex nature of environmental systems and the limitations of our analytical tools. We use heuristic
simulation examples to show how environmental heterogeneity and limitations in scientists' ability to measure and model environmental systems can give rise to conflicting inferences, even for very simple systems. We suggest that multiple studies with conflicting findings often provide an opportunity to expand and refine our understanding of complex environmental systems: conflicting findings from different studies often represent different partial views of a complex system. Using hierarchical Bayesian techniques, we show how pooling multiple studies can enable development of a richer understanding of SES dynamics, and better estimation of uncertainties. These findings suggest that resolving disputes centered around conflicting information often (but not always) requires integration of information, rather than adjudication of "correct" and "incorrect" findings.

Chapter 3 focuses on the prediction and avoidance of ecological regime shifts. Recent research indicates that changes in ecosystem dynamics, such as increased variability, could potentially serve as early warning indicators of impending shifts (Kleinen et al. 2003; Carpenter and Brock 2006; van Nes and Scheffer 2007; Guttal and Jayaprakash 2008). A critical question is whether such indicators provide sufficient warning to adapt management to avert regime shifts. We examine this issue using a fisheries model, with regime shifts driven by angling (amenable to rapid reduction) or shoreline development (only gradual restoration is possible). We find that if drivers can only be manipulated gradually, management action needs to be taken substantially before a shift to avert it; if drivers can be rapidly altered aversive action may be delayed until a regime shift is underway. Large increases in regime shift indicators only occur once a shift is initiated, often too late for management to avert a shift. To be useful in averting regime shifts, we conclude that
research needs to focus on defining critical levels of the regime shift indicators. Ideally, these critical levels should be related to switches in the ecosystem attractors — the point at which the long-term dynamics of the ecosystem switch to a different regime. In the chapter, we develop and present a new spectral density ratio indicator to this end.

Chapter 4 investigates factors that may enable transformation in ecosystem management approaches. Many authors contend that transformation of the largely sectoral, expert-centered ecosystem management institutions of modern, Western societies, to more integrated, collaborative, adaptive ecosystem governance approaches would improve our ability to sustainably manage ecosystems in support of human well-being (Cortner & Moote 1999; Berkes et al. 2003; MA 2005). We investigate transformation in ecosystem management as a process of social innovation, focusing specifically on factors that may facilitate the generation and adoption of new approaches to ecosystem management. We base our findings on analyses of three local-level case studies involving transformation in freshwater governance. We suggest that ongoing environmental degradation, increasing environmental awareness, and shifting societal values are creating fertile ground for the emergence and adoption of new approaches to ecosystem governance. Our findings indicate that initiatives which foster environmental awareness and attachment to local ecosystems, develop environmental leadership capacity, and promote dialogue between key stakeholders may greatly facilitate the emergence of integrated, collaborative institutions. Our findings also underline the importance of institutional support in sustaining newly emerged collaborative governance institutions.
Chapter 5 focuses on how the mindsets and tools needed to address the challenges posed by the 21st century might be taught at the graduate level. Today's major environmental issues are complex and characterized by high-stakes decisions and high levels of uncertainty. While traditional scientific approaches are essential, contemporary challenges also require new tools and new ways of thinking. We provide an example of how such "post-normal" scientific approaches (Funtowicz and Ravetz 1993) have been taught through practical application of scenario planning (van der Heijden 1996; Peterson et al. 2003). The insights presented in this chapter are based on the experiences of students who participated in a one-credit graduate seminar on scenario-planning in the Spring of 2007. As part of the seminar, we developed four scenarios for the future of Lake Wingra in Madison, Wisconsin, in collaboration with a local community group, the Friends of Lake Wingra. Students found the course highly stimulating, thought-provoking and inspiring. Key learning points were recognizing the need for multiple points of view in understanding complex environmental issues, and better appreciating the pervasiveness of uncertainty. Students found collaboration with non-academic stakeholders particularly insightful and enjoyable. Importantly, most students left the course feeling more positive and inspired about the potential contribution they can make to addressing the environmental challenges society faces.

Conclusion

The frameworks by which we understand and study the world fundamentally shape and constrain the solutions we seek to the environmental challenges society faces. The development and adoption of a complex systems view is revolutionizing the way we understand ecosystems, and enabling us to see new possibilities for how we might manage
ecosystems in the face of escalating human demands. However, this new understanding also emphasizes that society's ability to understand and control ecosystems is limited. To sustain the ecosystems on which society depends, society needs to become adept at living with these evolving systems, rather than seeking to command them. This change in the perceived relationship between people and nature is already producing a vast array of institutional and technological innovations relating to the management of ecosystems. With sufficient political will and some luck, a wave of these innovations will become mainstream, transforming society in ways that improve human well-being now and in the future.

References


CHAPTER 2

SPURIOUS CERTAINTY:

HOW IGNORING MEASUREMENT ERROR AND ENVIRONMENTAL
HETEROGENEITY MAY CONTRIBUTE TO ENVIRONMENTAL CONTROVERSIES

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Abstract

Disputes over conflicting information from different studies play a prominent role in many environmental controversies. Rather than resulting from flawed or fraudulent science, we suggest that conflicting findings may often result from the complex nature of environmental systems and the limitations of our analytical tools. Using heuristic simulation examples we show how environmental heterogeneity and limitations in our ability to measure and model environmental systems can give rise to conflicting inferences even for very simple systems. Although analytical tools are improving, individual studies will remain limited in their ability to describe complex environmental systems. Instead of being a curse, we suggest that multiple studies with conflicting findings often provide an opportunity to expand and refine our understanding of complex environmental systems. Using hierarchical Bayesian techniques, we show how pooling multiple studies can enable us to generate a broader, richer, and more accurate understanding of the dynamics of a system and to better estimate our uncertainties.

Key words: Hierarchical Bayes, meta-analysis, measurement error, environmental heterogeneity, complex systems
Introduction

Many environmental disputes ostensibly center on conflicting findings reported by different scientific studies. The current controversy over marine fisheries management, for example, involves a heated debate about the extent to which marine fish stocks are overfished (reviewed in Hilborn 2007). When conflicting results emerge from different studies it is often alleged that they are due to flawed or fraudulent science on the part of one or more parties (Herrick and Jamieson 2001; Sarewitz 2004; Bocking 2004). Such allegations tend to fuel and polarize the controversy. They often also lead us to attempt to resolve the conflict by trying to establish who is right and who is wrong. What is seldom recognized is that allegations of poor science implicitly assume that an observable and largely invariable set of processes underlie the disputed issue. But what if the processes being studied vary over time or space or are not readily observable? In the complex systems that form the setting for most environmental disputes this is often the case.

Complex systems, such as biological and social systems, are characterized by high dimensionality and the ability to self organize and adapt (Manson 2001; Gunderson & Holling 2002; Liu et al. 2007). This means that complex systems consist of many, highly connected components that are constantly changing in response to changes in the external environment, as well as in response to evolution and change within the system itself. In contrast to the immutable laws that govern many individual physical and chemical processes, important relationships within a complex system therefore tend to vary over space and time (Gunderson & Holling 2002; Taylor 2005). An important consequence of this variability is that a study at one time and place may observe a different set of processes, or relative
importance of different processes, than a comparable study at another time and place. Where variability over time or space has a regular pattern and occurs at a high enough rate, it may be possible to quantify this variability (Hilborn & Mangel 1997; Clark 2007). However, when the variability in a process does not have a regular pattern or occurs very slowly, it may be difficult to detect or understand.

To further complicate matters, important aspects of complex systems may not be directly observable. Environmental scientists often have to rely on surrogate measures to make inferences about the true variables of interest (Hilborn & Mangel 1997; Clark 2007). For instance, knowledge of the average concentration of phosphorus in a lake over the summer season is important for lake management (Carpenter 2003) but this aggregate measure cannot be observed directly. Instead average seasonal concentrations have to be inferred from individual samples taken at specific locations at specific points in time. Another manifestation of non-observability is that manipulative studies are often not possible at the scale of the systems of interest. Environmental scientists often rely on observational studies where the causal impacts of different factors cannot be clearly isolated (Hilborn & Mangel 1997; Gelman & Hill 2007). For instance in the estuary of a large river, a decline in the fish catch may be observed at the same time as an increase in the number of fisherman, the amount of shoreline development, and the level of effluent discharge. This can lead to considerable uncertainty about the true impact of particular factors and how different casual factors interact with one another.
Difficulties in understanding the dynamics of complex systems are being dealt with in at least two ways. First, much effort is being invested in improving the quantitative tools for measuring and analyzing complex systems. Advances in computing power over the past two decades have revolutionized our quantitative tools and ability. Of particular relevance to environmental sciences are hierarchical Bayesian techniques (Wade 2000; Ellison 2004; Clark 2005). Hierarchical Bayesian approaches enable us to better represent the nature of complex environmental systems: they enable us to better account for interrelationships across space and time, and for multiple sources of uncertainty (e.g. in predictor as well as response variables) and variability (e.g., within lakes as well as across lakes) (Clark 2005; Clark 2007).

Second, and somewhat paradoxically, there are increasing suggestions that the nature of complex systems may make their dynamics inherently uncertain and unpredictable, no matter how sophisticated our tools become (Zellmer et al. 2006; Pahl-Wostl 2007; Pilkey & Pilkey-Jarvis 2007; Roe and Baker 2007). Accordingly, effort is being invested in thinking about complex systems in alternative ways that rely more strongly on qualitative approaches, or combinations of qualitative and quantitative approaches, such as integrated environmental assessments (Farrell & Jäger 2005; Mitchell et al. 2006). These approaches typically draw on a diverse range of perspectives and interpretations in trying to understand the different ways in which a complex system may function or develop over time. At the same time, these approaches acknowledge that the dynamics of the system may never be fully knowable, and are crafted to function on the basis of preliminary, best available information.
This paper sets out to demonstrate two things in the context of environmental conflicts: 1) The inherent character of the complex systems environmental scientists study, combined with the limits of our statistical tools, mean that different studies may often produce conflicting results. Such conflicting findings may often be best interpreted as representing different partial views of a system, rather than some findings being correct and others wrong. 2) Cooperative approaches that focus on synthesizing information from multiple studies may be a good strategy for resolving environmental controversies. Not only do such approaches help build trust and common understanding, but they also enable us to broaden our understanding beyond that provided by individual studies. We use simple simulation examples to illustrate two specific mechanisms that can give rise to conflicting inferences: i) measurement error, and ii) spatial/temporal variability. We then show heuristically how pooling information from different studies using hierarchical Bayesian techniques can expand and refine our understanding and help synthesize conflicting inferences. Since many readers may not be familiar with hierarchical Bayesian techniques or the impacts of measurement error, we start with a brief overview of each before turning to the simulation examples.

Hierarchical Bayesian approaches enable more complex representations of systems

Hierarchical Bayes is one of several new tools that enhance our ability to analyze the dynamics of complex environmental systems. Bayesian statistics represent an alternative approach to widely-used classical or frequentist statistical techniques. The philosophical differences between Bayesian and frequentist statistics, and their implications for
environmental science have been discussed in detail (e.g., Howson and Urbach 1991; Wade 2000; Ellison 2004). We do not repeat these here, but rather highlight key features of hierarchical Bayes (as distinct from simple Bayes (Clark 2005)) that make it particularly useful for analyzing complex environmental systems and informing policy. These features include the ability to decompose complex problems into computable submodels within a coherent, consistent framework, to account for stochasticity in multiple components and at multiple levels of a system, to pool diverse data sources, and to enable better assessment of low probability parameter values.

Hierarchical modeling enables complex problems to be represented and analyzed as a series of simpler conditional models (box 1) (Wikle 2003; Clark 2005; Clark & Gelfand 2006). The Bayesian framework is highly suited to hierarchical approaches, although hierarchical modeling can also be carried out in non-Bayesian frameworks (Raudenbush & Bryk 2002; Gelman & Hill 2007). An important advantage of fully Bayesian approaches are that inference about every parameter takes into account the uncertainty about all other parameters. In contrast, frequentist, maximum likelihood and empirical Bayesian approaches are based on large sample theory, and inferences about key parameters do not fully account for uncertainty about unknown auxiliary parameters (Carlin & Louis 2000; Raudenbush & Bryk 2002). Fully Bayesian approaches therefore provide more precise estimates of stochasticity in a system, particularly when sample sizes are small, as is often the case at higher levels of hierarchical models (Raudenbush & Bryk 2002; Gelman & Hill 2007).
Two types of stochasticity are important for statistical inference and prediction: variability and uncertainty (Hoffman and Hammonds 1994; Ferson and Ginzburg 1996; Clark 2005). Variability is inherent to many aspects of complex systems, and refers to fluctuations or variation in the system, for instance in the daily discharge of a particular river. Variability does not decline with increasing sample size, it merely becomes better quantified. In contrast, uncertainty relates to our knowledge of parameters that describe the system, for instance the mean and variance in daily river discharge. Uncertainty about system parameters tends to decline asymptotically as sample size increases. The simulation examples presented in this paper show how the hierarchical Bayesian framework allows the basic linear model (Model A) to be expanded to include uncertainty (measurement error) in the explanatory variables (Model B), and variability in the process parameters (Model C) (Figure 1).

Hierarchical Bayesian approaches are particularly useful in providing a framework for pooling multiple data sources. Environmental scientists are frequently confronted with a diverse array of data relevant to a particular problem, such as data from geographical information systems, field experiments and expert surveys, often representing processes operating at different spatial or temporal scales (Wikle 2003). The Bayesian framework provides a means of coherently integrating such differing types of information. The hierarchical nature of the framework avoids the need to disaggregate or aggregate data to a common scale of analysis (e.g., in a study of multiple lakes, precipitation measured at a regional weather station can be incorporated at the regional scale instead of the common practice of assuming the same regional precipitation value for each lake, as if precipitation had been measured at the lake level). Inappropriate aggregation and disaggregation of data
can bias inference (Osborne 2000; Raudenbush & Bryk 2002). Statistical meta-analyses, which synthesize experimental results from multiple studies, are a technique that are based on this capability for hierarchical synthesis (Fernandez-Duque and Valeggia 1994). In this paper, we use hierarchical Bayes to pool data from multiple studies in order to more accurately describe the relationships in a system, and better quantify variability and uncertainty.

An important feature of hierarchical Bayesian inference is that unknown parameters are assumed to be random variables drawn from specific probability distributions, rather than fixed unknown quantities (Ellison 2004; Clark 2005). The output of a Bayesian analysis therefore consists of probability distributions for the unknown parameters (the so-called posterior distribution), rather than point estimates and confidence intervals. The posterior distribution allows assessment not only of the most likely value of a parameter based on the sample data, but also values reflected in the tails of the distribution that, although unlikely, are consistent with the data. This information may be highly relevant to assessing risk, particularly if there is a potential threshold at which impacts are greatly increased.

**Measurement error may bias regression results**

Measurement error arises when field measurements are imprecise or when a surrogate measure is used to represent the true variable of interest (Fuller 1987; Carroll et al. 1995; Gustafson 2004). The observed variable represents the true variable plus some error. Measurement error is present in all environmental field studies, but the degree of error depends on the characteristics of the variable being investigated. Some phenomena may be
highly variable, rare, or difficult to measure accurately and precisely given available measurement techniques, making them more prone to measurement error. For example, spatial clustering characteristically results in substantial measurement error in fish stock assessments (Hilborn & Walters 1992) and censuses of birds and large mammals (Gaston and McArdle 1994).

Measurement error is therefore related to sampling and variability. In many cases, the variables of interest in a statistical analysis represent averages or other aggregate measures of individual observations. For example, $X$ may be a random variable representing the true mean length of all adult trout in a particular stream. $X^*$ may represent an estimate of $X$ based on a sample of 15 adult trout. For small sample sizes and high variability among individuals, we would expect that $X^*$ would often “measure” $X$ with a substantial degree of error.

Using variables measured with error to infer empirical relationships with other variables can result in biased conclusions. In the classical linear model, the response variable $Y$ is assumed to be a random variable whose values follow a specific probability distribution (e.g., normal), and measurement error is accommodated by the $\varepsilon$-term (Figure 1, Model A). The explanatory variables $X$ are, however, assumed to be fixed variables with specified values that are measured with no error (Sokal & Rohlf 1995; Quinn & Keough 2002). This assumption holds in situations such as designed experiments where $X$ represents fixed treatment levels set by the researcher. However, in many environmental science applications $X$ represents a random variable subject to measurement error, just like $Y$. In fact, $X$ and $Y$ are pairs of observations $(X,Y)$ that represent a sample from the joint distribution of the two
variables. An appropriate model for incorporating variability in $X$ is Model B (Figure 1). However, because Model B is not readily accommodated within the classical statistical framework, it is common practice within the environmental sciences to assume that Model A is a reasonable simplification (Quinn & Keough 2002).

Depending on the objective of the analysis, assuming Model A instead of Model B may or may not be appropriate (Quinn & Keough 2002). If the aim of the analysis is prediction, using Model A with noisy data $X^*$ in place of $X$ will not impact the ability to predict $Y$. In essence, the analysis establishes a relationship between $X^*$ and $Y$, and this will be adequate if inference or future predictions are to be based on data $X^*$ with the same level of measurement error as that in the data used to calibrate the model. However, if the aim of the analysis is to describe the true nature of the relationship between $X$ and $Y$ (i.e., to estimate the true $\beta$), assuming Model A may not be appropriate. In environmental controversies, establishing the true relationship between $X$ and $Y$ often becomes important in order to, for instance, quantify the degree to which degradation in a response $Y$ is due to the activities $X$ of specific groups. Accurately quantifying the true relationship is also important in more mechanistic models, where the degree of measurement error in the calibration data $X^*$ may often differ from that of the simulated prediction data. In simple linear regression, if measurement error is present in a single predictor $X$ but not accounted for, the analysis will underestimate the true strength of the relationship between $X$ and $Y$ (Fuller 1987; Carroll et al. 1995). Box 2 demonstrates this effect.
The impacts of measurement error in linear regression analysis are worsened if two or more correlated explanatory variables are present. The presence of multiple explanatory variables that are related to one another is common in the complex settings that characterize environmental disputes. Where one of two correlated explanatory variables is measured less precisely than the other, the attenuation of the relationship between that variable and the response will be amplified. At the same time, the estimated strength of the relationship between the more precisely measured variable and the response will be inflated. These effects become more marked the higher the correlation between the explanatory variables. Figure 3 depicts this effect for the multiple linear regression model described in box 3.

As shown in the next section, it is possible for studies that use data measured with differing degrees of precision to reach opposite conclusions about the relative importance of different explanatory factors. Although a substantial body of literature exists on correcting for measurement error (e.g., Fuller 1987; Carroll et al. 1995; Gustafson 2004), in practice the level of measurement error in environmental data is usually unknown and uncorrected for. Quantifying the degree of error is costly as it requires replicate measures of the variables measured with error, or that the relationship between the true and surrogate measures of a variable be established through a validation study. The statistical skills required to correct for measurement error are also more advanced. Consequently, unless it has been established that measurement error is important in a particular context, environmental analyses generally assume measurement error is negligible (Quinn & Keough 2002).
Although we focus on the simplest case of a two-predictor linear regression with additive measurement error, measurement error is also important in linear models with more than two explanatory variables, as well as in non-linear models. In linear models where more than two predictors are measured with error, the impacts of measurement error become less predictable and depend on the degree of correlation between the variables and the extent of measurement error in each explanatory variable (Gustafson 2004). In non-linear models, such as logistic models, and for more complex measurement error structures, such as measurement errors that are correlated with errors in the regression model, attenuation or inflation of the estimated regression coefficients may or may not occur, depending on the specific model and the nature of the measurement error (Fuller 1987; Carroll et al. 1995). Importantly, the degree and direction of bias may vary over the sample and parameter space (Stow and Reckhow 1996).

Conflicting inferences: Two examples

We use two simple simulation examples to demonstrate how different research groups may arrive at conflicting conclusions about the relationships between variables due to i) differences in measurement error in the data they collect, or ii) environmental variability. We also show that by pooling data from different studies using a hierarchical Bayesian approach, we may gain a better understanding of the true relationship between the variables and the uncertainty about our inferences. We use a two-predictor linear process because it represents the simplest model that allows us to demonstrate these ideas.
**Example 1: Conflicting results due to measurement error.** Imagine three studies are conducted to establish the relationship between some response variable \( Y \) and two explanatory variables \( X_1 \) and \( X_2 \). For instance, some early studies of the causes of freshwater eutrophication used time series data of chlorophyll \( a \) (\( Y \)), phosphorous (\( X_1 \)) and nitrogen (\( X_2 \)) seasonal mean concentrations to establish the relative contribution of nitrogen and phosphorous to algal blooms (Schindler et al. *in press*). In these types of data sets, phosphorous and nitrogen concentrations are usually correlated.

For the purpose of the simulation example, assume that the true relationship between \( Y, X_1 \), and \( X_2 \) is as described in box 3. To investigate the impact of measurement error on inference in the three studies, we assume that only \( X_1 \) is measured with error, as \( X_1^{*} \) (box 3). We further assume that the true \( X_1 \) is the same in all studies (e.g., representing mean summer phosphorous concentration in particular lake over a series of years), and that all three studies use the same set of precisely measured data points for \( Y \) and \( X_2 \). Such a situation could arise if all studies were, for instance, conducted on the same lake, and each study collected their own data on \( X_1 \), but obtained precisely measured data on \( Y \) and \( X_2 \) from a central agency. Such “duplication” in data collection often occurs when the causes of environmental change are hotly disputed and different interest groups launch their own investigations, or when multiple agencies monitor the same ecological variables in a particular location. We assume that the degree of measurement error \( \tau \) in \( X_1 \) is different in each study: For study 1 assume that \( \tau = 10 \), for study 2 that \( \tau = 2 \), and for study 3 that \( \tau = 5 \). Such differences in measurement error may arise if, for example, the \( X_1^{*} \) data used by each study were collected using a different method (perhaps related to differing budgets for data collection), or if the
measurements $X_i^*$ depended on weather conditions and these were different when each study collected their data (e.g., phosphorous levels may be elevated directly after a thunderstorm, and measurements taken at such times may impact seasonal mean estimates).

As might be done in the field, assume that each study collects a set of 50 data points $(Y,X_1^*,X_2)$ and estimates $\beta_0, \beta_1$ and $\beta_2$ assuming Model A (Figure 1), using either classical or Bayesian approaches. The measurement errors for the three simulated studies correspond to CVs in the observed $X_i^*$ of 17%, 11% and 11% respectively. Importantly, substantial differences in measurement error therefore do not necessarily translate into big differences in the observed CVs between studies. In practice this means that one cannot easily tell whether a particular study contains substantial measurement error compared to another study simply by comparing the spread of the data.

Figure 4 gives the results of the three imagined studies. While study 1 finds that $X_2$ has a larger impact on the response $Y$ than does $X_1$, study 2 arrives at the opposite conclusion. Meanwhile study 3 finds that $X_1$ and $X_2$ have approximately the same effect on $Y$. These differences in the estimated parameters are entirely due to differences in the observed $X_i^*$ resulting from the differences in measurement error $\tau$ between the three studies. The $r^2$ statistics, which reflect the proportion of the variance in $Y$ explained by the models, are 0.78, 0.96 and 0.81 for the three respective studies. For the Bayesian analyses, a measure of fit is provided by the Deviance Information Criterion (DIC), where lower values of DIC indicate a better fit. The DIC values for the three studies are 562, 474 and 554 respectively. Although study 2 is clearly superior, all three studies provide a reasonable fit to the data. If these
models have very different policy implications, where some policies impose much higher costs on a particular societal sector than others (e.g., the extent to which phosphorous versus nitrogen is regulated in order to manage eutrophication), these differences may not be sufficient to ward off intense controversy.

Next, we show that it may be possible to help resolve conflicting inferences by pooling the data from the different studies and applying the more complex Model B (Figure 1). As in this example, data collected by different studies can sometimes be regarded as replicate measures of the same underlying variable, allowing one to estimate and correct for measurement error. Figure 4 shows the results of applying Model B to the pooled data. The $r^2$ for the pooled model, calculated as defined by Gelman and Hill (2007), is 0.99 and the DIC for the response $Y$ is 377. With only three replicates it is not possible to fully assess and correct for measurement error, but it is clear that pooling helps synthesize the results from the different studies and provides a more accurate assessment than at least two of the three studies. In the absence of clear reasons for choosing amongst models, or when political reasons make it difficult to do so, pooling information that can be regarded as different measures of the same underlying variable may often present a good option. The more replicate studies available, the better measurement error can quantified and corrected for.

A more insidious problem may arise if multiple studies all use data with approximately the same degree of measurement error. In such cases there will be strong agreement between the different studies, but their conclusions may all be erroneous. As an example, assume that instead of three studies, there exist ten studies which all collect data $X_i^*$ with measurement
error $\tau = 10$ for the same system described in box 3. We assume again that the true $Y$, $X_1$, and $X_2$ are the same in all studies. As shown in Figure 5, there will be a high level of agreement between the studies but all will substantially underestimate the true impact of $X_1$. The strong agreement between studies may lead to misplaced confidence in the findings. Where these studies can be regarded as replicates, they can be pooled in order to estimate and correct for measurement error by applying Model B (Figure 1). This results in a pooled estimate $\hat{\beta}_1$ that differs substantially from that of any of the individual studies (Figure 5). These results suggest that even where different studies do not produce conflicting findings, combining the information from multiple studies may produce insights not evident from the individual studies.

In situations where measurement error may be significant, but the information in different studies cannot be regarded as replicate measures of the same underlying variable, alternative techniques are required to estimate and correct for measurement error. The advantage of the approach presented here is that where studies can be regarded as replicates, it offers a way to draw on existing data sources to estimate and correct for measurement error.

**Example 2: Conflicting results due to environmental heterogeneity.** Assume as before that three studies collect data in order to establish the relationship between $Y$ and two explanatory variables $X_1$ and $X_2$. In this example we assume that no measurement error is present in any of the data. Instead, we examine the impact of the true process $\beta_1$, varying between studies. In the eutrophication example introduced above, this could happen if the different studies are conducted on different lakes, and the strength of the process $\beta_1$ (the
relationship between chlorophyll $a$ and phosphorous concentration) is somewhat different in each of these locations due to for instance differences in zooplankton grazing.

For the simulation model, we again assume that the true process is as described in box 3, except that instead of the relationship $\beta$, being fixed, it varies over time or space such that $\beta_i \sim n(10,4)$. The actual $\beta_i$, underlying the data observed in each study is assumed to represent one random sample from this distribution. In addition, given the context that each study is conducted in a different location we assume that the 50 observations $(Y_i, X_i, Z_i)$ by each study represent independent random samples, i.e. the true $Y$, $X$, and $X^2$ are different in each study.

Figure 6a presents the findings from the three simulated studies, where the data from each study is analyzed assuming Model A (Figure 1). While each of the three studies accurately represent the strength of the relationship $\beta$, as expressed at the particular time and location of their study, there may be substantial differences in the findings between studies. It is not unusual to find substantial variability between studies, for instance in plant or vertebrate abundances under particular land uses (Scholes and Biggs 2005). Where such conflicting results are the only information available for general policy decisions and hold high stakes for certain interest groups, they may lead to intense controversy. In a sense, each of the individual studies produces spurious confidence in the generality of their findings.

One strategy for better capturing variability is to expand the spatial or temporal scope of individual studies. For instance, studies could be conducted over multiple years, or across
different types of ecosystems. Funding and logistical constraints, however, often limit the degree to which individual studies can capture the broad range of variability that characterize many environmental systems. By enabling the pooling of multiple datasets, approaches such as hierarchical Bayes can provide a practical mechanism for synthesizing the variability reflected in the findings of disparate individual studies.

Figure 6 presents the estimated mean ($\hat{\theta}$) and standard deviation ($\hat{\lambda}$) of the process $\beta_i$, as estimated by pooling the three studies using a hierarchical Bayesian approach and assuming Model C (Figure 1). Each study is assumed to represent one sample of the true process $\beta_i \sim n(0, \lambda)$. As is evident from Figure 6, our knowledge about the true process $\beta_i$ may be highly uncertain if the number of individual studies is small. Without pooling, substantial uncertainties about the true process $\beta_i$ may be hidden by the high degree of certainty at the level of the individual studies.

**Expanding our view**

Disputes over factual information play a prominent role in many environmental controversies. This situation is partly a consequence of modern societies’ reliance on scientific evidence as the basis for environmental decision-making (Cortner & Moote 1999; Sarewitz 2004; Bocking 2004). When information becomes available that threatens the interests of particular groups, they are motivated to scrutinize the information or provide counterevidence, often by commissioning their own studies. For many environmentally contentious issues, this response tends to result in a proliferation of scientific studies (Jasanoff and Wynne 1998; Sarewitz 2004). However, rather than resolving disputes and
enabling decision-making, multiple studies often produce conflicting findings and lead to growing political controversy and gridlock (e.g., in cases such as climate change, nuclear waste disposal, and endangered species) (Jasanoff and Wynne 1998; Sarewitz 2004). While scientific information clearly has a critical and valuable role to play in formulating environmental policy, it is increasingly evident that traditional science alone can seldom resolve contentious environmental debates, particularly in the short to medium term when policy action is often least costly and most beneficial. The examples in this paper provide a heuristic for understanding how multiple studies might lead to conflicting insights due to factors such as measurement error and environmental heterogeneity.

Different studies commonly frame and investigate problems in different ways. A substantial body of literature documents how scientific knowledge is never independent of political context, but rather is co-produced by scientists and the society in which they are embedded (Ziman 2000). Different ways of framing, investigating and interpreting issues, associated with diverse values and interests, usually results in differing, but equally legitimate findings (Jasanoff and Wynne 1998; Sarewitz 2004). From this perspective, conflicting findings may often be best regarded as representing different partial views of a rich and complex reality, rather than some findings being “correct” and others “incorrect”. However, the traditional scientific process does not provide a ready mechanism for uniting disparate partial views into a coherent whole that can serve as a basis for policy action (Ziman 2000; Sarewitz 2004; Bocking 2004). Instead, different interest groups typically select from the myriad partial views those that support their interests, and the debate becomes gridlocked in technical disputes over the validity of different findings. As pointed out by Sarewitz (2004), scientific
uncertainty in this sense may be understood not so much as a lack of scientific understanding, but rather as a lack of coherence among competing scientific understandings.

An important characteristic of partial views is that, by limiting their focus, they tend to underestimate the uncertainties associated with their findings. As illustrated by the examples in this paper, where only a single study is available it may create a misplaced sense of confidence about our knowledge of a particular system. This confidence may be reinforced by political processes that demand clear-cut scientific evidence to support policy action (Bocking 2004). Overconfidence in the results of particular analyses can have major implications for assessing policy options and risks. For example, the collapse of the multi-billion dollar Newfoundland cod fishery in Canada in the early 1990s was largely a result of overly liberal fishing quotas derived from stock models in which fishing mortality was underestimated. Although several reports appeared in the late 1980s suggesting that fishing mortality was being underestimated, they were downplayed by the administration in an attempt to maintain constancy and the credibility of the management agency’s models (MacGarvin 2001).

Our findings suggest that the conventional view that duplication of scientific studies is undesirable, and that conflicting results are a curse to effective environmental management, is often misguided in the context of complex environmental systems. As demonstrated by our heuristic examples, multiple studies often provide an opportunity to gain a broader perspective and to develop a more nuanced and rich understanding of the dynamics of environmental systems. Multiple studies may also highlight uncertainties in our knowledge
that may be hidden in individual studies. Although conflicting results are often seen as creating uncertainty, they may be better interpreted as making us aware of our uncertainties – in effect, making unknown unknowns known. Such knowledge may help guard against placing undue faith in our understanding of a system and becoming overly narrow or arrogant in the management strategies we pursue. In this respect, conflicting results may often represent a blessing rather than a curse and encouraging multiple, diverse perspectives can help better inform environmental management.

In order to realize the value of multiple studies, mechanisms are needed for synthesizing disjointed information. The examples in this paper demonstrate one information-pooling tool, hierarchical Bayes, which enables quantitative synthesis of multiple studies. Better understanding environmental systems through pooling approaches requires the same careful thought and evaluation that characterizes all good science. Quantitative information pooling techniques should not be treated as a panacea where information can be mindlessly tossed together (Eysenck 1994). Although flawed or disingenuous science is relatively uncommon and usually detected by the scientific process (Herrick and Jamieson 2001), it does occur and simply pooling such information with legitimate information will degrade the quality of the synthesis. Tools such as hierarchical Bayes are best used to synthesize diverse findings from what are believed to be good scientific studies, rather than as a means to weed out bad science.

Several additional information pooling approaches, which go far beyond the quantitative techniques illustrated in this paper, have developed over the past two decades. The most
prominent example of these new approaches that combine qualitative and quantitative synthesis techniques are large integrated environmental assessments to deal with issues such as ozone depletion, climate change and loss of ecosystem services (Farrell & Jäger 2005; Mitchell et al. 2006). Other examples include adaptive ecosystem management (Holling 1978; Walters 1986; Walters and Holling 1990), scenario planning (van der Heijden 1996; Peterson et al. 2003) and joint fact finding (Andrews 2002; Schultz 2003; Karl et al. 2007). These approaches all involve pooling diverse information and building common understanding (among scientists, as well as between scientists and policy-makers or ecosystem managers) about the dynamics and uncertainties of complex problems. The examples in this paper provide a metaphor for how such information pooling approaches can expand our understanding substantially beyond that provided by individual studies. When employed together with consensus-building techniques, these approaches can provide a mechanism for developing a coherent, broad and widely agreed upon perspective as a basis for policy action. The Intergovernmental Panel on Climate Change is a prime example of how such information pooling approaches have helped synthesize diverse partial perspectives on a highly complex issue, and helped advance policy action.

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Box 1. Hierarchical Bayesian inference

Hierarchical modeling is based on a simple rule of probability: The joint probability distribution \( p(W,X,Y) \) of a collection of random variables \( W, X \) and \( Y \) can be factorized into a series of conditional distributions, i.e., \( p(W,X,Y) = p(Y|W,X) \times p(W|X) \times p(X) \). A conditional distribution, such as \( p(W|X) \), is the probability distribution of a random variable \( W \) given that \( X \) is known.

This rule provides a mechanism for deriving a complex joint distribution from a series of relatively simple conditional models. Specifying the joint distribution of a complex spatio-temporal process such as a forest landscape, such that it captures stochastic behavior at multiple spatial locations across time, may be very difficult or impossible to do directly. However, the joint distribution can be derived by specifying the distribution of the relevant conditional models (e.g., tree structure is conditional on climate, land use, fire, etc), which is often a much easier task (Wikle 2003).

The aim of Bayesian inference is to estimate the probability that a hypothesized process (e.g., linear \( Y = \beta X + \epsilon \)) with specific parameter values is true given the observed data (Ellison 2004). This so-called posterior probability can be calculated by decomposing the problem into several components (Wikle 2003; Clark 2005):

\[
p(\text{process, parameters} | \text{data}) \propto p(\text{data} | \text{process, data parameters}) \times p(\text{process} | \text{process parameters}) \times p(\text{all parameters})
\]

[Data model] \hspace{1cm} [Process model] \hspace{1cm} [Parameter model]
The data model is also known as the likelihood and the parameter models as priors (or hyperpriors). The process model can be considered part of the likelihood or the prior, depending on context.

As an example, the stochastic linear process \( Y = \beta X + \varepsilon \), where \( \varepsilon \sim n(0, \sigma^2) \) denotes a normal distribution with a mean of zero and a variance equal to \( \sigma^2 \) (Figure 1, Model A), can be decomposed as:

\[
p(\beta, \varepsilon, \sigma | X, Y) \propto p(Y | X, \beta, \varepsilon) \times p(\varepsilon | \sigma) \times p(\beta) \times p(\sigma)
\]

An advantage of the hierarchical framework is that it can be readily extended to incorporate additional sources of uncertainty or variability. For example, if the process \( \beta \) varies over time or space such that \( \beta \sim n(\theta, \lambda) \) (Figure 1, Model C) this can be included as:

\[
p(\beta, \theta, \lambda, \varepsilon, \sigma | X, Y) \propto p(Y | X, \beta, \varepsilon) \times p(\beta | \theta, \lambda) \times p(\varepsilon | \sigma) \times p(\theta) \times p(\lambda) \times p(\sigma)
\]

The framework can also readily accommodate multiple data sources. For example, if two datasets \((X_1, Y_1)\) and \((X_2, Y_2)\) are available to estimate Model A (Figure 1) these can be incorporated as:

\[
p(\beta, \theta, \lambda, \varepsilon, \sigma | X_1, Y_1, X_2, Y_2) \propto p(Y_1 | X_1, \beta, \varepsilon) \times p(Y_2 | X_2, \beta, \varepsilon) \times p(\varepsilon | \sigma) \times p(\beta) \times p(\sigma)
\]

An example of integrating data collected at different scales is provided by Ladeau and Clark (2006). These authors used a hierarchical Bayesian approach to estimate the effect of elevated \( \text{CO}_2 \) on tree fecundity, individual variability in fecundity, and interannual variability in fecundity by integrating cone count data from individual trees with plot-level seed-trap
data. An example of integrating data collected at multiple sites is given by Reckhow (1996), who assessed trends in biological integrity in the Scotio River in Ohio. Reckhow shows how a hierarchical approach can be used to draw on collateral information from other sites to improve site-specific estimators of biological integrity, especially where sample sizes are small.
Box 2. Measurement error impacts in simple linear regression

The impact of measurement error in simple linear regression is best illustrated by a simple simulation example. Assume that two random variables $X$ and $Y$ are related to one another by $Y = \beta X + \varepsilon$, and that true $X \sim n(0, 1)$, true $\beta = 1$, and $\varepsilon \sim n(0, 0.5^2)$. Using this information, we can generate data pairs $(X, Y)$ that represent the true variables of interest.

Now assume that we have a sample of 50 data pairs $(X, Y)$, as though they had been precisely measured in the field. Analogous to the approach many scientists would adopt, this data can be used to derive an estimate $\hat{\beta}$ of the true relationship $\beta = 1$ using simple linear regression. The results of such an analysis are depicted in Figure 2a.

To demonstrate the impact of measurement error on the estimate $\hat{\beta}$, assume that the true $X$ and $Y$ are measured imprecisely as $X^*$ and $Y^*$ respectively. We assume measurement error in the form of simple additive noise, i.e., $X^* = X + \nu$ and $Y^* = Y + \nu$, where $\nu \sim n(0, 1.5^2)$. Figure 2b depicts the results of a simple linear regression analysis based on the dataset $(X, Y^*)$ and Figure 2c depicts the results based on the dataset $(X^*, Y)$. In the latter case it is clear that assuming Model A (Figure 1) to be an adequate description of reality when the data are in fact generated according to Model B (Figure 1) results in an underestimate of the strength of the true relationship ($\beta$) between $X$ and $Y$.

A Bayesian representation of these findings is given by plotting the distribution of the estimates $\hat{\beta}$ derived from 5000 simulations of the analyses presented in Figures 2a-c. The estimated $\hat{\beta}$ is slightly different in each simulation due to different random samples of $X$ and $Y$. 
Box 3. Multiple linear model example

In the controversy examples developed in this paper, we assume that the true response $Y$ is related to two explanatory variables $X_1$ and $X_2$ by the multiple linear model

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \epsilon$$

We assume that the true $X_1$ and $X_2$ are $n(100, 10^2)$ random variables drawn from a multivariate normal distribution with correlation $\rho = 0.5$, and that $\epsilon \sim n(0, \sigma^2)$. For the purpose of the examples, we assume that the true $\beta_0 = 25$, $\beta_1 = 10$, $\beta_2 = 5$, and $\sigma = 10$. Since $X_1$ and $X_2$ have the same mean and variance, their direct relative impacts on the response $Y$ can be assessed by comparing the coefficients $\beta_1$ and $\beta_2$.

In the examples where measurement error is present (Figures 3-5), we assume that $X_2$ is measured precisely, while the true $X_1$ is measured imprecisely as $X_1^* = X_1 + \nu$ where $\nu \sim n(0, \tau^2)$

In the example dealing with environmental heterogeneity (Figure 6) we assume that instead of $\beta$, assuming a single fixed value, it varies over time and/or space such that $\beta_i \sim n(0, \lambda)$.

The Bayesian analyses assume $n(0, 100^2)$ priors for the unknown $\beta$ parameters and the unknown true $X_i$, and uniform densities $u(0, 100)$ for the standard deviation parameters $\sigma$, $\tau$ and $\lambda$. As recommended by Gelman (2006) and Gelman and Hill (2007) we do not use inverse-gamma prior distributions due to their sensitivity to standard deviations close to zero.
**Figure 1.** Hierarchical Bayesian approaches provide a coherent framework for including multiple sources of uncertainty and variability in a system. The classical linear regression model (Model A) assumes that the unknown process parameter $\beta$ has a single fixed value and that unexplained random variability ($\varepsilon$) is associated only with the $Y$ variables. A Bayesian framework allows us to expand this model to, for instance, include random variability in the observed explanatory variables $X^*$ in the form of measurement error ($\nu$) (Model B), or to model variability in the process $\beta$ (Model C). Unboxed variables/parameters are assumed to have fixed values, boxed variables/parameters are assumed to be random variables with specific probability distributions (e.g., normal), and the shaded boxes represent unexplained random noise in the models.
Figure 2. (a) The estimated relationship $\hat{\beta}$ between the true $X$ and $Y$, based on 50 data points measured without error (numbers in parentheses indicate two standard errors). (b) Measurement error in the response $Y$ does not substantially affect the magnitude of the estimate $\hat{\beta}$; it simply increases the uncertainty about the estimate. (c) In contrast, measurement error in $X$ substantially attenuates the strength of the estimated relationship $\hat{\beta}$ between $X$ and $Y$. In addition, the confidence interval around $\hat{\beta}$ remains narrow, and we are therefore spuriously confident about our estimate. (d) Changes in the estimated $\hat{\beta}$ are also apparent in the posterior probability distributions derived from a Bayesian analysis: measurement error in $Y$ results in a flattening of the estimated distribution of $\hat{\beta}$, whereas measurement error in $X$ results in the distribution shifting toward zero.
Figure 3. (a) Where measurement error is present in only one of two explanatory variables ($X_i$ with error $\tau$), the estimated coefficient $\hat{\beta}_i$ is attenuated. (b) In addition, if $X_i$ and $X_2$ are correlated ($\rho \neq 0$), the coefficient $\hat{\beta}_2$ is inflated despite the fact that $X_2$ is measured with no error. A similar result holds when the true $\beta_2 = 0$, i.e. $X_2$ is a spurious variable that has no real impact on the response $Y$. Results are based on 100 multiple linear regression analyses at each level of measurement error in $X_i$. 
Figure 4. Studies can arrive at conflicting conclusions entirely as a consequence of differences in measurement error in the analyzed data. Here, studies 1, 2 and 3 reach very different conclusions about the strength of the coefficient $\beta_1$ compared to $\beta_2$ entirely as a consequence of differences in measurement error $\tau$ in the variable $X_r$. The unshaded bars in the upper panel show the estimated coefficients derived using classical multiple linear regression (error bars indicate two standard errors). The thin lines in the lower panel give the distribution of the estimated coefficients derived from a Bayesian analysis of the same data, uncorrected for measurement error (i.e., assuming Model A). By pooling the studies, it is possible to apply Model B (Figure 1) using a Bayesian approach and estimate and correct for measurement error (shaded bars and thick lines). Note that the pooled estimate does not simply represent the mean of the three studies.
Figure 5. (a) A high level of agreement between individual studies does not necessarily mean their findings are converging on the true ecosystem dynamics. In this example, ten individual studies observe variable $X$, with the same degree of measurement error $\tau = 10$ and analyze the data assuming Model A (Figure 1). Despite the high level of agreement between studies, all substantially underestimate the true $\beta_i$ (unshaded bars, with two standard errors). Pooling the ten individual studies in a Bayesian framework can enable one to estimate and correct for measurement error by applying Model B (shaded bar, with 95% Bayesian credible interval).

b) As the number of individual studies which are pooled increases, the pooled, corrected estimate $\hat{\beta}_i$ approaches the true $\beta_i$. The uncertainty about the estimate $\hat{\beta}_i$ (shown by the 95% credible interval) also decreases as the number of studies which are pooled increases.
Figure 6. (a) Differences between individual studies may also arise because the true process \( \beta_i \) varies over time or space. Individual studies may not be able to capture variability in \( \beta_i \), but instead represent individual samples of the process \( \beta_i \) at a particular time and place (unshaded bars, with the true \( \beta_i \) for each study indicated above). By pooling multiple studies we can attempt to estimate the mean of the process \( \beta_i \), as well as quantify our uncertainty about the mean (shaded bars with 95% credible intervals). (b) Similarly, pooling allows us to estimate the variability in the process \( \beta_i \). Variability of a process and uncertainty about this variability is often as or more important to decision-making than knowledge of the mean. (c) The estimated distribution of \( \hat{\beta}_i \) converges on that of the true process \( \beta_i \) as the number of studies that are pooled increases. When the total number of studies being pooled is small, adding additional studies can lead to dramatic improvements in estimation of \( \hat{\beta}_i \) (e.g., \( n = 3 \) versus \( n = 5 \)).
CHAPTER 3

TURNING BACK FROM THE BRINK: CAN AN IMPENDING REGIME SHIFT BE DETECTED IN TIME TO AVERT IT?

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Abstract

Ecological regime shifts are large, abrupt, long-lasting changes in ecosystems that often have considerable impacts on human economies and societies. Avoiding unintentional regime shifts is widely regarded as desirable, but prediction of ecological regime shifts is notoriously difficult. Recent research indicates that changes in ecosystem time series (e.g., increased variability and autocorrelation), could potentially serve as early warning indicators of impending shifts. A critical question is whether such indicators provide sufficient warning to adapt management to avert regime shifts? We examine this question using a fisheries model, with regime shifts driven by angling (amenable to rapid reduction) or shoreline development (only gradual restoration is possible). The model represents key features of a broad class of ecological regime shifts. We find that if drivers can only be manipulated gradually management action is needed substantially before a shift to avert it; if drivers can be rapidly altered aversive action may be delayed until a regime shift is underway. Large increases in the regime shift indicators only occur once a shift is initiated, often too late for management to avert a shift. To improve usefulness in averting regime shifts, we suggest that research focus on defining critical indicator levels rather than detecting change in the indicators. Ideally, critical indicator levels should be related to switches in ecosystem attractors; we present a new spectral density ratio indicator to this end. Averting ecological regime shifts will also depend on developing policy processes that enable society to respond more rapidly to impending regime shifts.
Introduction

Ecological regime shifts are large, sudden changes in ecosystems that last for substantial periods of time (Scheffer et al. 2001; Carpenter 2003). Regime shifts are a source of growing concern as rising human impacts on the environment are increasing the likelihood of ecological regime shifts at local to global scales (Steffen et al. 2004; MA 2005; IPCC 2007). Accumulating evidence suggests that regime shifts can occur in diverse ecosystems: they have been documented in oceans (Knowlton 1992; Hare and Mantua 2000; Jackson et al. 2001), freshwaters (Scheffer 1997; Carpenter 2003; Scheffer and van Nes 2007), forests (Ludwig et al. 1978), woodlands (Dublin et al. 1990), drylands (Foley et al. 2003), rangelands (Walker et al. 1981; Laycock 1991; van de Koppel et al. 1997) and agroecosystems (Cramer and Hobbs 2005; Anderies et al. 2006; Gordon et al. 2008). Regime shifts are widely regarded as undesirable as they often have considerable impacts on human well-being. For example, the collapse of Canada’s Newfoundland cod fishery in the early 1990s directly affected the livelihoods of some 35,000 fishers and fish-plant workers, led to a decline of over $200 million dollars per annum in revenue from cod landings (DFO 2004), and had significant indirect impacts on the local economy and society (Finlayson and McCay 1998). Ecological regime shifts are also undesirable because they may be very costly or impossible to reverse (Meijer 2000; Scheffer et al. 2001; Carpenter 2003). Regime shifts entail changes in the internal dynamics and feedbacks of an ecosystem that often prevent it from returning to a previous regime, even when the driver that precipitated the shift is reduced or removed (Scheffer et al. 2001; Carpenter 2003). For instance, despite a moratorium on the Canadian cod fishery for over 15 years, the fishery has shown little sign of recovery (DFO 2004).
Avoiding unintentional regime shifts is widely regarded as desirable (MA 2005; IPCC 2007). However, ecological regime shifts are notoriously difficult to predict. Most regime shifts come as surprises, and the conditions and mechanisms leading to them only become clear once the shift has occurred (Scheffer et al. 2001; Carpenter 2003). Regime shifts typically result from a combination of gradual changes in an underlying driving variable (or set of variables), combined with an external shock such as a storm or fire (Beisner et al. 2003; Scheffer and Carpenter 2003). Gradual changes in underlying drivers usually have little or no apparent impact up to a certain point, and then unexpectedly lead to a regime shift when that threshold is crossed. Once an ecosystem is close to a threshold, a shift is often precipitated by a shock that under previous conditions had no dramatic consequences (Scheffer et al. 2001; Carpenter 2003). Slow underlying drivers that push ecosystems towards thresholds often go unnoticed and are frequently associated with increased economic benefits. In the absence of known thresholds and impacts, it is very difficult to constrain such drivers. Rising demands on the world's ecosystems are therefore expected to increasingly push ecosystems towards ecological thresholds (Steffen et al. 2004; MA 2005; IPCC 2007). To avoid large-scale disruptions to human societies, there is accordingly an urgent need to improve our ability to anticipate and avert ecological regime shifts.

The need to better anticipate regime shifts has sparked much recent research (Kleinen et al. 2003; van Nes and Scheffer 2003; Rietkerk et al. 2004; Oborny et al. 2005; Carpenter and Brock 2006; van Nes and Scheffer 2007; Carpenter et al. 2008; Dakos et al. in press; Guttal and Jayaprakash 2008). Most of this work has been based on mathematical models of ecosystems with multiple stable states (Holling 1973; May 1977). The focus has been on
generic changes in system behavior that might enable one to predict regime shifts across a
diverse range of ecosystems, rather than experimental or model-based methods that focus on
better understanding the mechanisms of particular regime shifts in specific ecosystems. It
turns out that although little change may be evident in the average condition of an ecosystem
as a regime shift is approached, there may be detectable changes in other properties of
monitoring data. Specifically, time series data may show increased variability (Carpenter and
Brock 2006), changes in skewness (Guttal and Jayaprakash 2008), higher correlation through
time (Kleinen et al. 2003; Dakos et al. in press), and slower rates of recovery from
disturbances (van Nes and Scheffer 2007) in advance of regime shifts. Such changes in the
ways ecosystems behave hold substantial promise as early warning indicators of regime
shifts, although there are as yet few empirical tests (Kleinen et al. 2003; Carpenter and Brock
2006; van Nes and Scheffer 2007; Dakos et al. in press; Guttal and Jayaprakash 2008).
However, given their potential use as early warning indicators, a critical question is: Would
such changes in ecosystem behavior provide sufficient advance warning to adapt
management to avoid a regime shift? Or are the lags and momentum of change in the
ecosystem so great that by the time these changes are detected the system is already
committed to a shift? Answers to these questions clearly impact whether attempts to employ
such early warning indicators are worthwhile, and whether use of these indicators may be a
viable strategy for avoiding undesirable regime shifts.

To examine these questions, we use a fisheries food web model to explore 1) how close an
ecosystem can get to an ecological threshold and still avert a regime shift by implementing
changes in management, and 2) which regime shift indicators might give warning before this
“point of no return”. Regime shifts in the model can be triggered by two mechanisms: a) angling, which is amenable to relatively fast manipulation through management action, or b) shoreline development, which can only be manipulated gradually. This enables us to investigate the potential avoidance of regime shifts for fast versus slow management variables. We use a modeling approach to explore the use of regime shift indicators under the most favorable possible conditions: where the signal from the indicators is not obscured by environmental variability and management changes can be implemented without delay. If the indicators provide insufficient warning under these conditions they hold little promise as tools for avoiding regime shifts in practice. On the other hand, if they perform adequately further investigations into the generality of the indicators, their detection in the field, and suitable policy instruments for management response are warranted.

The model

The fisheries food web model is derived from the literature (Carpenter and Brock 2004; Carpenter et al. 2008), and involves a trophic triangle where adult piscivorous fish (A) prey on planktivorous fish (F), which in turn prey on juvenile piscivores (J) (Figure 1). The model includes movement between refugia and foraging arenas by planktivores and juvenile piscivores (Walters & Martell 2004). Two possible regimes exist: a piscivore-dominated regime and a planktivore-dominated regime. Nonlinear shifts between the regimes are precipitated by harvesting of adult piscivores (qE) or through shoreline development. Shoreline development impacts refuge habitat (fallen trees in shallow water), and hence affects the rate at which juvenile piscivores move from the foraging to the refuge arenas (b).
This in turn impacts the predation of juvenile piscivores and affects recruitment. See the Methods section for further details.

We employed two management policies in our simulations: 1) Immediate reduction in harvest to a level of $qE = 0.1$ (policy MS1), and 2) Gradual restoration of shoreline habitat, such that $h = +0.01$/year (policy MS2). In reality, political tradeoffs and bureaucratic delays mean that management responses will usually be substantially weaker and tardier than MS1 and MS2. In accordance with our aims, we chose optimistic management scenarios to explore the use of regime shift indicators under highly favorable conditions.

Results

We defined a regime shift as the period over which the annual increase in the planktivore ($F$) population exceeded 10%. In the model, regime shifts have a typical duration of approximately 15 years, reflecting plausible limits on growth rate of $F$.

*How close to a regime shift can the ecosystem get and still avert a shift?* Dramatic differences are evident in the proximity to a regime shift that can be reached where management action focuses on angling (MS1) as opposed to shoreline development (MS2). For the angling-induced regime change, a regime shift can be well underway (10 years into the shift) and a permanent change still averted by reducing harvest as per scenario MS1. In contrast, for a regime shift driven by shoreline development, habitat restoration according to scenario MS2 has to be initiated at least 45 years before the onset of the shift to avert it (Figure 2). Under these management options, avoiding a regime shift driven by shoreline
development therefore requires much earlier detection and action than avoiding a shift driven by angling. Sensitivity analyses show that the attainable proximity to a regime shift is reduced if the management response is weaker or if the system is subject to greater levels of stochastic variation, but is unaffected by the rate of change in the variable driving the regime shift (Supporting Information (SI) Figure S1).

We also explored manipulation of angling to avert a shoreline development-driven shift, and shoreline restoration to address an angling-induced shift. For the shoreline-driven shift, harvest-reduction policy MS1 (with $b$ fixed at its level at the time of policy implementation) enabled a much greater proximity to the regime shift (8 years into the shift) than policy MS2 (45 years before the shift). A proximity equivalent to MS1 alone was attained if MS1 and MS2 were implemented simultaneously. Analogous results were found for the angling-driven shift and a regime shift driven by concurrent changes in $qE$ and $h$ (SI Table S1). The attainable proximity to a regime shift is therefore determined by the nature of the management variables: where pressure on the ecosystem can be immediately reduced (angling) a shift can be averted by taking action even once a regime shift is underway, whereas if pressure can only be gradually reduced (restoration of shoreline habitat) action needs to be taken substantially in advance of a shift. The remainder of our analyses focus on the scenarios in Figure 2, as these capture the essential differences between fast and slow management variables in the model.

Two further features are evident in Figure 2. First, there is a very clear and abrupt end to the window in which management action to avert a regime shift is possible. For instance, for the
angling-induced regime shift, management action MS1 in year 91 leads to a rapid recovery of the piscivore population. However, the same management action just a year later is unable to prevent the regime shift, and the piscivore population never recovers despite a prolonged reduction in harvest.

Second, in transient settings such as those characterized in Figure 2, the actual regime shift occurs substantially later than the switch to the alternate attractor. Consequently, there is a considerable lag between the point at which the long-term sustainable level of \( q_E \) or \( h \) necessary to maintain a particular regime is surpassed, and the point at which a regime shift occurs. For instance, in the case of shoreline development the critical level at which the lower planktivore attractor disappears is \( h = 3.7 \). However, it is not until 80 years later when \( h \) has been reduced to \( h = 2.1 \) that a regime shift occurs. This may lead to the mistaken inference that the long-term sustainable level of \( h \) is close to the level at which the regime shift is initiated. More encouragingly, such lags provide room for “overshoot error”: it is possible to substantially exceed long-term critical levels and still avert a regime shift. For instance, while the critical long-term harvest level is \( q_E = 1.78 \), there is a window of 90 years, during which \( q_E \) can be further increased to as much as 2.23, when it is still possible to avert a regime shift by implementing response MS1.

*Which indicators provide warning before the “point of no return”?* Large annual increases in variance, skewness and kurtosis, and the AR1 coefficient of within-year planktivore population only occur once a regime shift is underway (Figure 3). In the case of the angling-driven shift and response MS1, such increases could provide warning 3-10 years
before the end of the policy window in year 91, depending on the sensitivity with which
“large annual increase” is defined. However, for a regime shift driven by shoreline
development with management restricted to MS2, large annual increases in the indicators do
not occur until several decades after the end of the policy window in year 35. “Spikes”
associated with regime shifts occur in the variance and AR1 indicators under a wide range of
model conditions. In contrast, skewness and kurtosis show spikes under some conditions,
but not others (such as high levels of environmental noise).

Prior to the onset of a regime shift, directional changes in angling or shoreline development
produce more gradual increases in variance and the AR1 coefficient (Figure 3). Such
increases could potentially provide greater advance warning of a regime shift. However, as
apparent in Figure 3, there are two difficulties associated with using gradual increases as early
warning indicators. First, it may be difficult to detect these changes above the inherent
variability in the ecosystem unless data were available for a substantial length of time. For
instance, no increase in variance or AR1 is detectable in the 30-year period (a considerable
time series in most real-world contexts) preceding the end of the policy window for the
shoreline-driven shift. Second, detecting a slow increase in variance or the AR1 coefficient
provides no information about how close to a regime shift the ecosystem may be. Gradual
increases in variance and AR1 simply indicate that there is ongoing, directional change in
some driver, and occur even when the driving variables are below the critical long-term
levels that would lead to a switch in the attractor. Detecting a gradual increase in variance or
AR1 therefore does not indicate whether or when management action is needed to avert a
regime shift.
To sustain desirable regimes, indicators of regime shifts should ideally provide warning when the long-term critical $qE$ or $b$ levels are being approached or have been exceeded — i.e. when a switch in the attractor occurs. No marked changes in variance, AR1, skewness, or kurtosis accompany a switch in the attractor (Figure 3). To investigate signals of a changing attractor, we calculated spectral densities based on the within-year planktivore data. We found clear signs of "reddening" after the switch from the lower to the upper attractor (Figure 4). Based on the spectra in Figure 4, we calculated a spectral density ratio to compare the contribution of low to high frequency processes to the total within-year variance in the planktivore population. In our model, the point at which low frequency processes start to dominate (10-year running mean spectral density ratio exceeds unity) provided warning of the disappearance of the lower attractor, for both the angling and shoreline development-driven shifts (Figure 5). The 10-year running mean threshold also performed well when the system was subject to larger amounts of stochastic variation, or when the rates of change in the driving variables were increased (SI Figure S2).

Discussion

Our model focuses on regime shifts precipitated by changes in slow underlying drivers, rather than shifts precipitated by large external shocks (Beisner et al. 2003; Scheffer and Carpenter 2003). Mathematically, regime shifts driven by slow underlying variables involve bifurcations: a small, smooth change in parameter values causes sudden qualitative changes in the long-term behavior of a dynamical system due to the appearance or disappearance of attractors. Ecological models have demonstrated that mathematical bifurcations provide a plausible explanation for regime shifts in a diverse range of ecosystems (Holling 1973; May
As human impacts on Earth expand, gradual changes in variables leading to bifurcations are likely to be a major cause of ecological regime shifts (Steffen et al. 2004; MA 2005; IPCC 2007). As in our model, some of these drivers will be amenable to fast manipulation through management action, whereas it will only be possible to manipulate others much more gradually. While not applicable to all ecological shifts, our model therefore represents key features of a broad class of potential regime shifts. For regime shifts driven by slow underlying variables, our findings relate specifically to: 1) the possibility of averting ecological regime shifts, and 2) the use of regime shift indicators to this end.

**Averting regime shifts.** Our findings emphasize the need for monitoring and proactive intervention in averting ecological regime shifts (MA 2005; Stern 2006; IPCC 2007), especially in cases where underlying drivers cannot be rapidly manipulated. Where it is possible to rapidly and drastically reduce impacts driving a shift (as in the case of angling), our results indicate that regime shifts could potentially be averted even once they are underway. However, bureaucratic inertia, policy compromise (Kingdon 1995), and the risk of unforeseen environmental shocks (Beisner et al. 2003; Scheffer and Carpenter 2003), make delaying action until a regime shift is underway a dangerous strategy even where it is theoretically feasible. If the variable driving a regime shift can only be manipulated gradually (as in the case of shoreline development), our results indicate that taking action substantially before the onset of a regime shift is crucial if a shift is to be averted. Proactive intervention is also desirable from the standpoint of cost, as the closer the system has moved to a regime shift, the stronger (and generally more costly and socially disruptive) the action needed to prevent a regime shift (Stern 2006; IPCC 2007).
Our results highlight that in systems subject to regime shifts there is often a discrete window for policy action, after which it becomes impossible to avert a shift. The existence of such windows, where the same action in two adjacent years could differ radically in its effectiveness, is seldom considered in environmental decision-making processes. Policy windows may help explain why some fisheries have shown rapid recovery when fishing controls were instituted while other fisheries, such as the Newfoundland cod, have shown little recovery despite prolonged reductions in harvest (Beddington et al. 2007; Hilborn 2007). Our findings also underscore the risks being taken by current inaction surrounding climate change. Atmospheric carbon dioxide ($\text{CO}_2$) levels are a variable that cannot be rapidly and drastically reduced. As highlighted by other authors, timely action to avert potential CO$_2$-induced regime shifts is therefore likely to be critical (Stern 2006; IPCC 2007).

Our results underscore the need for developing alternative decision-making processes for systems subject to ecological regime shifts (Norberg & Cumming 2008; Waltner-Toews et al. 2008). In transient settings, such as those that characterize our simulations and most real-world ecosystems (Holling 1973; Hastings 2004), long-term sustainable levels of human impact can easily be exceeded. Switches in system attractors will usually occur substantially before any noticeable effects on ecosystems become evident. By the time adverse environmental effects become apparent it is often too late to avert a regime shift. Trial-and-error approaches that wait for evidence of negative environmental impacts before taking action are therefore ill-advised. However, as evidenced by the issue of climate change justifying restraint of human impacts without evidence of negative environmental effects
represents a considerable challenge (Stern 2006; IPCC 2007). In such contexts, leading indicators could be a useful tool for policymakers.

In practice, regime shifts often involve multiple driving variables (Carpenter 2003; Kinzig et al. 2006). Where some of these variables can be manipulated quickly and others only more gradually, it may be possible to “buy time” to ameliorate a slow variable by implementing changes in a variable that can be rapidly manipulated. For instance, if the policy window for preventing a shoreline-induced regime shift through habitat restoration has closed, it may still be possible to avert a shift by reducing angling pressure. This effectively provides additional time to rehabilitate shoreline habitat. Identifying fast management variables that can act as “emergency levers” to extend our opportunity for addressing slow management variables may be critical to avoiding regime shifts. In the case of climate change, such actions could potentially include activities such as large-scale reforestation programs that can help provide the time needed to restructure energy systems (Stern 2006; IPCC 2007).

Use of regime shift indicators. Regime shift indicators need to be further developed and refined if they are to detect impending shifts with sufficient warning to avert regime shifts. Rapid increases in variance, skewness and kurtosis, and the AR1 coefficient only occur at the onset of a regime shift. In most cases such increases occur too late for management action to avert a shift. While gradual increases in variance and AR1 occur substantially prior to a shift, these trends are problematic as early warning indicators. First, because the changes are small, detection is only likely in variables monitored over substantial periods of time, where the rate of change in the driving variables is high, or if baseline data from a much earlier period in
time were available. Second, even if an increase in variance or AR1 is detected, it provides no indication of how close to a regime shift the ecosystem is, or even whether the long-term sustainable levels of the underlying drivers have been exceeded or not.

To provide useful early warnings, it is necessary to determine specific values of the regime shift indicators that should trigger management action, rather than simply detect trends. Ideally, critical warning levels should be related to the point at which switches in the attractor occur, since this is the impact level of concern for longer-term sustainability. We have attempted to define such an indicator by means of a spectral density ratio. For our model, it appears that a shift from dominance by high frequency processes (spectral ratio < 1) to dominance by low frequency processes (spectral ratio > 1) provides a robust indicator of a switch in the attractor. These ratios need to be tested across a broader set of models and field data to assess the degree to which they are ecosystem-specific. Critical indicator levels could also be defined in terms of variance or AR1, but our analyses suggest that such critical levels will tend to vary with environmental conditions.

An advantage of defining critical levels in absolute terms is that the possibility of impending regime shifts can then be assessed from intensive time series data collected over a relatively short period of time. With the rapid growth in high frequency environmental monitoring equipment, intensive time series are becoming available for a wide range of ecosystems (DeFries and Townshend 1999; Jeong et al. 2006). Nevertheless, determining critical indicator levels may be challenging and will need to draw heavily on model simulations, experimental manipulations and long-term observation of particular ecosystems. Given
research constraints, the degree to which critical levels turn out to be ecosystem-specific will largely determine their potential use for helping avert regime shifts. However, it may be possible to select or define indicators that have critical levels which are relatively transferable between ecosystem types or subtypes. This is an important area for future research.

In conclusion, our results indicate that regime shift indicators cannot at present be relied upon as a general means for detecting and avoiding ecological regime shifts. It is as yet unclear whether use of regime shift indicators for this purpose is achievable, but to the extent that it may be, our work suggests that it would rely on: 1) defining critical levels of the regime shift indicators, 2) linking these critical levels to long-term sustainable impact levels, and 3) finding or developing indicators that have critical levels that are relatively transferable across different ecosystem types. In addition, our results highlight the need for research on policy processes that are better suited to managing complex systems subject to regime shifts. Such processes would reduce inertia and enable society to respond more rapidly to information about impending regime shifts, better account for the existence of policy windows when planning management interventions, and rely on leading indicators, rather than adverse environmental impacts, as triggers for management action. While this research develops, management for unwanted regime shifts will depend on existing approaches that hedge, avoid risk, maintain ecological resilience, or build social resilience to cope with unexpected change (Carpenter 2003; Norberg & Cumming 2008; Waltner-Toews et al. 2008).
Methods

Model specification. Changes in the populations of adult piscivores (A), planktivores (F) and juvenile piscivores (J) are modeled on two times scales (Carpenter and Brock 2004; Carpenter et al. 2008). The dynamics over the shorter “monitoring interval” (taken as $1/50^{th}$ of a year) are given by:

\[
\frac{dA}{dt} = -qEA \\
\frac{dF}{dt} = D_F(F_R - F) - c_{FA}FA + \sigma \frac{dW}{dt} \\
\frac{dJ}{dt} = -c_{JA}JA - \frac{c_{FJ}vFJ}{h + v + c_{FJ}F}
\]

with parameters catchability ($q$), effort ($E$), exchange rate of $F$ between the foraging arena and a refuge ($D_F$), refuge reservoir of $F$ ($F_R$), consumption rate of $F$ by $A$ ($c_{FA}$), additive noise ($\sigma$), control of $J$ by $A$ ($c_{JA}$), consumption rate of $J$ by $F$ ($c_{FJ}$), rate at which $J$ enter the foraging arena ($\theta$), and rate at which $J$ seek refuge ($h$). The harvest rate is the product $qE$ and $dW/dt$ is a Wiener stochastic process.

Dynamics over the longer “maturation interval” (nominally one year; may be longer for species with slower maturation) are given by:

\[
A_{t+1} = A_{t+1;1} = s(A_{t,n} + J_{t,n}) \\
F_{t+1} = F_{t+1;1} = F_{t,n} \\
J_{t+1} = J_{t+1;1} = fA_{t+1}
\]
where \( t \) in \( A_{eq} \) denotes the maturation interval and \( n \) denotes the monitoring interval. \( s \) is survivorship between maturation intervals and \( f \) is the fecundity rate of \( A \). Parameter values are given in Table S2, and are based on the literature and whole lake experiments (Carpenter & Kitchell 1993; Carpenter et al. 2001; 2008). Phase diagrams for equilibrium conditions under different combinations of \( qE \), \( b \) and initial \( A \) are given in Figure S3.

**Simulations.** Our simulations involve transient conditions and were initiated near steady-state values corresponding to \( qE = 1.5 \) and \( b = 8 \) and run for a burn-in period of 500 years, before slowly increasing \( qE \) (at a rate of +0.005/year) or decreasing \( b \) (at a rate of -0.02/year until \( b = 0 \)) over an additional 500 year period. These parameter values were chosen for illustrative purposes; other values gave analogous results. Simulations were performed in R (R Development Core Team 2006).

**Regime shift indicators.** All indicators were calculated from the 50 simulated “monitoring” data points for \( F \) within each year. We calculated variance, skewness, and kurtosis using the R Moments package (R Development Core Team 2006), and mean-detrended OLS autoregressive lag 1 (AR1) coefficients, and the spectral density ratio using R functions. We did not use return time to equilibrium since our simulations involve ongoing changes in \( qE \) and \( b \). The spectral density ratio is derived from spectral densities based on an AR1 fit to first-difference detrended \( F \) data. AR1-based spectra are a standard tool in time series analysis (Chatfield 1989; Wei 1990). We defined the spectral density ratio as the ratio of the spectral density at a frequency of 0.05 (low) to the density at a frequency of 0.5 (high).
Acknowledgements

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References


Figure 1. The fisheries food web model used to explore the use of regime shift indicators in averting ecological regime shifts. Regime shifts are driven by angling or shoreline development. Angling directly affects the populations of adult piscivores through harvesting ($q_E$). Shoreline development affects the amount and quality of refuge habitat, and thus the rate at which juvenile piscivores hide from predators ($h$).
Figure 2. The attainable proximity to a regime shift is much greater for driving variables that can be rapidly manipulated (angling) than for variables that can only be manipulated gradually (shoreline development). Year 0 is defined as the year in which the switch from the lower \( F = 0.34 \) to the upper \( F = 76.92 \) planktivore attractor occurs. (A) In the absence of policy action, a harvest-driven shift occurs in years 81 to 95. The switch in attractors occurs at \( qE = 1.78 \). (B) The window for averting a regime shift by implementing harvest-reduction policy MS1 lasts to year 91, well within the regime shift. (C) However, instituting MS1 just a year later cannot avert a regime shift. (D) In the absence of policy action, a shoreline development-driven shift occurs between years 80 and 95, and switch in attractors occurs at \( h = 3.7 \). (E) To avert a regime shift, shoreline restoration policy MS2 has to be implemented substantially before the shift, by year 35. (F) Taking action slightly later (year 37) cannot avert a regime shift, although it is substantially delayed.
Figure 3. Large annual increases in the regime shift indicators only occur once a regime shift is underway, which is often too late for management action to avert a shift. Gradual increases in variance (A, D) and AR1 (C, F) may occur prior to a regime shift, but specific thresholds need to be defined to indicate whether or when policy action to avert a shift is required. Indicators are based on the within-year planktivore data and correspond to the changes depicted in Figs. 2A and 2D (vertical lines show the end of the policy windows).
Figure 4. For the angling-induced regime shift in Figure 2A, clear signs of spectral “reddening” are evident after the switch from the lower to the upper planktivore attractor. The spectral density describes how variation in the within-year planktivore population data may be accounted for by cyclic components of different frequencies as determined by Fourier analysis. Each line gives the AR1-based spectral density for one year. Analogous spectra exist for the shoreline-driven regime shift.
Figure 5. The point at which the spectral density switches from domination by high to domination by low frequency processes (10-year running mean spectral density ratio exceeds 1) provided warning of a shift in the attractor in our model. We defined the spectral density ratio as the ratio of the spectral density at a frequency of 0.05 (low) to the density at a frequency of 0.5 (high) as given in Figure 4.
Supporting online material

Table S1. Regardless of the underlying driver of the regime shift, manipulation of the variable amenable to rapid change (angling, via MS1) enabled much greater proximity to a regime shift than the variable that could only be manipulated gradually (shoreline habitat, via MS2). Where MS1 was implemented for the shoreline-driven shift, \( h \) was fixed at the level it had reached at the time of implementation; similarly for \( qE \) where MS2 was implemented for the angling-driven shift, or where the shift was driven by concurrent changes in \( qE \) and \( b \). HMS2 depicts a hypothetical policy where shoreline habitat is instantaneously restored such that \( b = 8 \). If this were this possible, a substantially greater proximity to the shoreline-driven shift would be attainable.

<table>
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<tr>
<th>Regime shift in the absence of policy</th>
<th>Angling-driven shift ((qE = +0.005/\text{yr}))</th>
<th>Shoreline-driven shift ((h = -0.2/\text{yr}))</th>
<th>Concurrent angling &amp; shoreline-driven shift</th>
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<td>Final year for policy:</td>
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<td>MS1</td>
<td>Year 91</td>
<td>Year 88</td>
<td>Year 76</td>
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<td>MS2</td>
<td>Year 17</td>
<td>Year 35</td>
<td>Year 14</td>
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<tr>
<td>MS1 and MS2</td>
<td>Year 91</td>
<td>Year 88</td>
<td>Year 76</td>
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<td>HMS2</td>
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Table S2. Definition of variables and parameters in the fisheries food web model. $A$ denotes adult piscivorous fish, $J$ denotes juvenile piscivores, and $F$ denotes planktivorous fish. Parameter values were taken from Carpenter et al. (2008). The symbol $T$ denotes the maturation interval and the symbol $t$ denotes the monitoring interval.

<table>
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<tr>
<th>Parameter</th>
<th>Value</th>
<th>Definition</th>
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<td><strong>Fixed parameter values (constant in all model runs)</strong></td>
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<td>0.3 $t^3 A^{-1}$</td>
<td>Consumption rate of planktivores by adult piscivores</td>
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<tr>
<td>$c_{JA}$</td>
<td>0.001 $t^3 A^{-1}$</td>
<td>Control of juvenile piscivores by adult piscivores</td>
</tr>
<tr>
<td>$c_{JF}$</td>
<td>0.5 $t^3 F^{-1}$</td>
<td>Consumption rate of juvenile piscivores by planktivores</td>
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<tr>
<td>$D_F$</td>
<td>0.1 $t^{-1}$</td>
<td>Exchange rate of planktivores between foraging and refuge arenas</td>
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<tr>
<td>$F_R$</td>
<td>100 $F$</td>
<td>Refuge density of planktivores</td>
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<tr>
<td>$v$</td>
<td>1 $t^{-1}$</td>
<td>Rate at which juvenile piscivores enter the foraging arena (vulnerability parameter)</td>
</tr>
<tr>
<td>$f_A$</td>
<td>2 $J A^{-1}$</td>
<td>Fecundity rate of adult piscivores</td>
</tr>
<tr>
<td>$A_0$</td>
<td>200 $A$</td>
<td>Initial adult piscivore population; ~equilibrium at $qE=1.5$, $h=8$</td>
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<tr>
<td>$s$</td>
<td>0.5 $T^{-1}$</td>
<td>Over-winter survivorship of adult and juvenile piscivores</td>
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<tr>
<td>$\sigma_F$</td>
<td>0.1 $F$</td>
<td>Standard deviation of the Wiener stochastic process</td>
</tr>
<tr>
<td><strong>Default values for parameters varied between model runs</strong></td>
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<tr>
<td>$qE$</td>
<td>1.5 $t^{-1}$</td>
<td>Harvest rate (catchability x fishing effort)</td>
</tr>
<tr>
<td>$h$</td>
<td>8 $t^{-1}$</td>
<td>Rate at which juvenile piscivores seek refuge (hiding parameter)</td>
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Figure S1. Sensitivity of the policy window to rate of change, strength of policy response and level of stochastic variability. (A) The angling-induced regime shift from Figure 2A used to compare the latest year in which policy action can avert a regime shift. Year 0 is the year in which the switch in the attractor occurs in the base scenario. (B) A greater rate of change in the driving variable \( qE \) leads to an earlier switch in the attractor (38 years before the base scenario), a regime shift earlier in time (years 14-29 in the absence of a policy intervention), and hence an earlier need to respond (by year 24). However, the opportunity for response is similar: up to 10 years into the regime shift. The greater rate of change in \( qE \) leads to a system that is further out of equilibrium, and hence greater overshoot of the critical \( qE = 1.78 \). (C) A weaker policy response (reducing fishing pressure instead of eliminating it) requires a policy response almost a decade earlier (by year 83 at the latest) to avoid a regime shift. (D) Substantially greater noise in the annual planktivore population (\( \sigma_F = 0.5 \)) only slightly reduces the possible window for policy response and the possible overshoot of \( qE \). Similar changes in \( h \) give analogous results.
Figure S2. Performance of the spectral density ratio under conditions of higher system variability and greater rates of change in the driving variables. Year 0 is the year in which the switch in the attractor occurs in the base scenario depicted in Figure 5 of the main text. (A,C) The point at which the 10-year running mean spectral density ratio exceeds 1 performs adequately as an indicator of a shift in the planktivore attractor under conditions of increased systems variability ($\alpha_c = 0.5$ as opposed to $\alpha_c = 0.1$) for both the angling and shoreline-driven shifts. Under conditions of higher variability, the policy windows are slightly reduced. (B) The spectral density indicator also performs adequately under a greater rate of increase in harvest levels ($qE = +0.015/\text{year}$ as opposed to $qE = +0.005/\text{year}$), or (D) a doubling of the rate of decrease in $b$ ($b = -0.04/\text{year}$ as opposed to $b = -0.02/\text{year}$). Policy windows are based on MS1 for the angling-induced shift and MS2 for the shoreline-driven shift.
Figure S3. Equilibrium phase diagrams for the anthropogenic drivers (A) angling and (B) shoreline development, derived from a deterministic version of the model. As $qE$ increases or $h$ decreases, the domain of attraction for the piscivore-dominated state declines. $qE = 0.54$ denotes the maximum sustainable yield level of harvest.
CHAPTER 4

NAVIGATING THE BACK LOOP: FOSTERING SOCIAL INNOVATION AND TRANSFORMATION IN ECOSYSTEM GOVERNANCE

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In review: Ecology and Society
Abstract

Addressing the environmental challenges of the 21st century requires substantial changes to the ways modern society views and manages ecosystems. In particular, many authors contend that fundamental transformation of the largely sectoral, expert-centered ecosystem management institutions of modern, Western societies is needed. There is increasing agreement that more integrated, collaborative, adaptive ecosystem governance approaches would improve society’s ability to sustainably manage coupled social-ecological systems. Accordingly, factors that enable transformation between sectoral, expert-centered and integrated, collaborative ecosystem governance approaches have become an active research area. In this paper, we investigate such transformations as a process of social innovation. Using three local-level case studies involving transformation in freshwater governance, we investigate factors that promote the emergence and spread of integrated, collaborative ecosystem governance approaches. Based on our investigations, we suggest that ongoing environmental degradation, increasing environmental awareness, and shifting societal values are creating fertile ground for the emergence and adoption of new approaches to ecosystem governance. Based on the case studies we examined, we suggest that initiatives which foster environmental awareness and attachment to local ecosystems, develop environmental leadership capacity, promote dialogue between key stakeholders, and provide institutional support to newly emerged institutions may greatly facilitate the emergence of integrated, collaborative governance approaches.

Key words: adaptive cycle, social innovation, transformation, ecosystem governance, complexity, bridging organization, co-management
Introduction

Many ecosystems, at scales ranging from local to global, are now heavily human-influenced, and undergoing large, far-reaching changes that threaten human well-being (Steffen et al. 2004; MA 2005; IPCC 2007). There is growing consensus that addressing this situation requires significant changes to the ways society views and manages ecosystems (Cortner & Moote 1999; MA 2005; Waltner-Toews et al. 2008). Specifically, it is recognized that the sectoral, expert-centered, command-and-control approaches that have largely characterized ecosystem management in modern, Westernized contexts over the past century are an important obstacle to addressing many contemporary environmental challenges. These expert-centered approaches view ecosystems as largely separate from human systems and typically aim to maximize human well-being by optimizing and stabilizing ecosystem outputs (Holling and Meffe 1996; Gunderson & Holling 2002; Walker & Salt 2006). Modern ecosystem management institutions associated with these approaches are largely predicated on the assumption that optimal management strategies can be and are best derived through apolitical scientific analysis (Cortner & Moote 1999; Bocking 2004).

It is now increasingly agreed that ecosystems are better viewed not as separate from human systems, but rather as coupled social-ecological systems (SES) that are continually evolving and changing (Berkes & Folke 1998; MA 2005; Norberg & Cumming 2008). Furthermore, it is becoming evident that the complex nature of SES pose inherent limitations on our ability to understand, predict and control SES (Pilkey & Pilkey-Jarvis 2007; Roe and Baker 2007), and that few, if any, scientific analyses are truly apolitical (Cortner & Moote 1999; Bocking 2004; Pielke 2007). Consequently, there is an emerging consensus that SES require
“governance” rather than simply “management” (Stoker 1998; Boyle et al. 2001): deciding on desired ecosystem outcomes, dealing with uncertainty and risk, and resolving trade-offs requires societal deliberation and cannot be decided through scientific analysis alone (Cortner & Moote 1999; Bocking 2004; MA 2005). Ecosystem management involves operationalizing such societal decisions (Boyle et al. 2001), which vary across cultures, locations and time. Given the complex nature of SES, it is increasingly agreed that ecosystem governance therefore needs to be adaptive, integrated (across disciplines and sectors), and polycentric (occurring at multiple levels), in addition to being tailored to particular contexts (Berkes et al. 2003; Dietz et al. 2003; Folke et al. 2005).

Many authors contend that transforming the existing largely sectoral, expert-centered ecosystem management institutions of modern Western states to realize more integrated, collaborative, adaptive forms of governance, at least at the local scale, is central to addressing the environmental challenges of the 21st century (Gray 1995; Cortner & Moote 1999; Senge et al. 2004; MA 2005). Encouragingly, examples of such transformation exist, largely at the local scale (e.g., Yaffee et al. 1996; Olsson et al. 2004). These examples provide valuable insight into the conditions and processes necessary to bring about transformation in ecosystem governance approaches. One way in which case studies of transformation in ecosystem governance have been analyzed is to identify different stages or phases in the process of transformation (Olsson et al. 2004; Olsson et al. 2006). Another way of analyzing such case studies, which we explore in this paper, is to view transformation in ecosystem governance as a process of social innovation. In the following section we provide a brief
overview of what social innovation is and how it relates to ecosystem governance transformation and the adaptive cycle (Holling 2001; Gunderson & Holling 2002).

The purpose of this paper is to apply a social innovation lens to explore factors that may help precipitate and facilitate successful transformation ("navigation of the back loop") from a sectoral, expert-centered approach to ecosystem management, to a more integrated, collaborative, adaptive form of ecosystem governance. Our approach is exploratory, and aims to provide a pilot assessment of the usefulness of applying a social innovation framework in understanding transformations in ecosystem governance, and to suggest factors that could form the basis for more in-depth inquiries. We focus specifically on elucidating factors that may promote bricolage and diffusion – the development and spread of social innovations in ecosystem governance. To investigate these factors, we explored three local-level case studies involving transformations in freshwater governance i) The Kristianstad Vattenrike wetlands in southern Sweden, ii) the Sabie River in eastern South Africa, and iii) The Yahara Lakes near Madison, Wisconsin in the USA. By employing a social innovation framework, we hope to be able to contribute new insights that complement existing work on bringing about transformations in ecosystem governance (e.g., Olsson et al. 2004; Loorbach 2007).

Social innovation and transformation

Social innovation refers to new concepts, strategies, initiatives, products, processes or organizations that meet pressing social needs and extend and strengthen civil society (The Young Foundation 2006; Westley et al. 2006). Social innovations can be pioneered by a wide
range of actors, including non-governmental organizations (NGOs), community groups, charities, governments, business, academics, philanthropists, or combinations of these groups (The Young Foundation 2006). Innovation differs from invention in that it does not refer only to the creation of new ideas or products, but also to the process of implementation or diffusion that make promising ideas useful in meeting social needs (The Young Foundation 2006; McKeown 2008). Any process of social innovation therefore consists of two dynamics: i) “bricolage”, or recombining existing and new ideas to form something novel (Levi-Strauss 1962), and ii) “contagion” or “diffusion”, the adoption and spread of novel ideas or inventions (Rogers 1995; Westley et al. 2005).

Two broad categories of innovation can be distinguished: incremental innovation and radical innovation (McKeown 2008). Most innovation is incremental and represents evolutionary, stepwise improvements to existing ideas, products or processes. Incremental innovation has a high chance of success and low uncertainty about outcomes. Radical innovation, on the other hand, involves the development and adoption of entirely new ideas, products or processes. It requires larger leaps of understanding and often new ways of seeing a problem. The chances of success are difficult to estimate and there is initially often considerable opposition to such ideas (The Young Foundation 2006; McKeown 2008).

Incremental and radical innovation in complex SES can be usefully understood in terms of the adaptive cycle (Holling 2001; Gunderson & Holling 2002) (Fig. 1). The front loop of the adaptive cycle can be seen as largely characterized by incremental innovation, while the back loop may be precipitated by or create a window of opportunity for radical innovation.
Changes and innovations that occur in the back loop may lead to fundamental reorganization of the SES, so that the system functions in a qualitatively different way than it did before – i.e. is transformed (Walker et al. 2004; Chapin et al. In press). The type of innovation primarily being called for in the context of ecosystem management, and the focus of this paper, is radical innovation. In the remainder of the paper, we use the term “innovation” to imply radical innovation.

Transformative social innovations often occur through waves of individual innovations. For instance, during the period of rapid industrialization and urbanization in 19th century Europe, a huge upsurge in social innovation occurred, leading to the development of microcredit facilities, building societies, cooperatives, and trade unions that collectively met an array of new social needs and helped transform societies (The Young Foundation 2006). A similar wave of innovation is needed if modern society is to transform its current ecosystem management institutions to meet the environmental challenges of the 21st century (MA 2005; Martin 2007; Waltner-Toews et al. 2008).

Much has been written on sources of innovation and enhancing the diffusion of innovations, especially in the business arena (von Hippel 1988; Rogers 1995; Fagerberg et al. 2006). Factors promoting social innovation are, however, relatively unstudied (The Young Foundation 2006). Available work suggests that social innovation is greatly facilitated by finance specifically directed at supporting innovation; incubation processes that nurture promising innovations in their early stages; leaders who visibly encourage and reward successful innovations; promotion of interactions across organizational, sectoral or
disciplinary boundaries; empowerment of users and stakeholders to drive innovation themselves; and opening of markets and governance processes to user groups and private businesses (The Young Foundation 2006). Importantly, social innovation cannot be directly planned and produced; it can only be stimulated by creating an environment conducive to innovation. Like any innovative process, rates of success can be increased, but substantial failure rates for social innovations are to be expected (The Young Foundation 2006; Westley et al. 2006; McKeown 2008).

**Methods**

The objective of this study was to explore factors that may facilitate transformation from sectoral, expert-centered approaches to ecosystem management, to more integrated, collaborative, adaptive forms of ecosystem governance, as viewed from a social innovation perspective. We used an exploratory case study approach (Yin 1994) as a suitable method for exploring factors that may facilitate transformation. Exploratory case study approaches are appropriate where i) the goal is to develop hypotheses and propositions for further inquiry, ii) it is not possible to control the situation being investigated, through for instance experimental manipulation, and iii) a holistic approach that considers the interplay of factors in the richness of contemporary real-world contexts is required to understand “how” and “why” certain events occurred (Yin 1994; Stake 1995). Given the exploratory approach, potential factors facilitating transformation were not predefined but rather allowed to emerge from the study. Importantly, results from exploratory case studies should not be taken as conclusive; detailed, rigorous explanatory case study approaches are needed to firmly establish causal connections between conditions and events (internal validity). The
results of this study therefore highlight topics for further research rather than providing conclusive findings.

We chose to compare multiple case studies in order to investigate commonalities and differences in the factors that may facilitate ecosystem governance transformation in different contexts. In case study research, multiple case study designs follow a replication, rather than a sampling, logic (Yin 1994); accordingly the three case studies were regarded as three replicate experiments. We were particularly interested in the extent to which common factors could be identified across the three case studies, as these would suggest important research topics for subsequent in-depth explanatory case studies. The understanding gained from such in-depth studies could inform policy strategies that may applicable in stimulating ecosystem governance transformation in other locations (Yin 1994; Miles & Huberman 1994). In order to broaden the potential applicability of our results (external validity), we selected case studies from three different regions of the world: Sweden, South Africa, and Wisconsin, in the United States. While all three cases represent modern Westernized ecosystem management contexts, their cultural and economic contexts differ markedly (Fig. 2). We suggest that common factors identified as facilitating transformation across these three cases may be applicable in other countries with relatively developed economies and strong European heritages. The extent to which these factors may hold in other contexts is a subject for further research. All three case studies focus on transformations in freshwater governance. The extent to which our findings may be applicable to governance transformations in other arenas, such as biodiversity or agriculture, cannot be judged from our case studies and needs to be elucidated through further research.
In addition to geographic diversity, we selected the case studies based on their representivity as examples of local-scale transformation in ecosystem governance, and the availability of information and interviewees. The criteria for judging whether each case study qualified as an example of local-scale transformation in ecosystem governance included: i) Whether a new named and identifiable entity (e.g., a bridging organization) emerged that linked previously disparate projects and groups and improved coordination in ecosystem management, ii) This entity existed for at least a decade, iii) The entity was deemed by most stakeholders to have had a significant impact in terms of improving local environmental conditions, or at least preventing further deterioration of environmental conditions, and iv) The domain of influence of the entity was local in scale: sub-national, and sub-state or sub-provincial level.

Case study approaches typically draw on a variety of information sources including interviews with actors, written records, and personal observation. We investigated the process of social innovation in each case study based on the published literature (Table 1) and in-depth, open-ended interviews (Rubin & Rubin 1995; Kvale 1996) with two to three people knowledgeable about each case. We interviewed both researchers and ecosystem managers in order to gain different perspectives on the factors important for transformation. The interviews were recorded for referral, and notes were made of points that were particularly emphasized by the interviewees. The interviews were conducted by asking interviewees to “tell the story” of how the new integrated, collaborative ecosystem management entity came to be. This enabled interviewees to relate their understanding of the connections between events from their own particular mental frameworks rather than
from the unfamiliar theoretical framework of social innovation. Interviewees were, however, asked to specifically highlight factors that were critical to i) coming up with the idea for the new entity, ii) establishing and maintaining the new entity, and iii) factors that provided important obstacles to the establishment and maintenance of the new entity. The key common factors that appeared to facilitate transformation across the different case studies were synthesized and inferred based on these interviews and the literature about each case study (Table 1). Where interviewees indicated an interest to review the findings from the study, draft copies of the paper were provided and their comments incorporated.

The case studies

In all three case studies we explored, transformations in ecosystem governance occurred through the establishment of bridging organizations during the late 1980s and early 1990s. Bridging organizations link actors at different levels and thereby facilitate inter-organizational collaboration between, for example, governmental agencies, non-governmental agencies, businesses, and community groups (Brown 1991; Westley and Vredenburg 1991; Olsson et al. 2007). The bridging organizations in the three case studies we explored were established to address ongoing degradation of local freshwater ecosystems and aimed to coordinate and integrate fragmentary individual ecosystem management projects. In all three cases, the changes that occurred constituted a substantial reorganization in ecosystem governance, and can be interpreted as examples of transformation in this respect (Walker et al. 2004; Chapin et al. In press). In this section we briefly describe the transformations that occurred in each case study. In the next section we analyze in detail the factors that appear to underlie innovation and transformation across the three studies.
Kristianstads Vattenrike, Sweden

Kristianstads Vattenrike constitutes an extensive area of wetlands on the lower Helgeä River surrounding the town of Kristianstad in southern Sweden. The wetlands provide valuable ecosystem services such as flood control, cultural and recreational values, and flooded meadows for grazing and haymaking. The wetlands also support a rich floral and faunal diversity (Olsson et al. 2004). Importantly, much of the wetland diversity is maintained by grazing and haymaking practices, making local people cultural stewards of the wildlife habitats (Nabhan 1997; Olsson et al. 2004).

Since the founding of the town in 1614, and especially after World War II, growing developmental pressures led to increasing degradation of the Kristianstad wetlands (Magnusson 1981). In an attempt to preserve the wetlands, a section of the area immediately adjacent to the Helgeä River became designated as a Ramsar Wetland of International Importance in 1975. This resulted in several conservation plans aimed at protecting the area from further exploitation. However, management of the wider Kristianstads Vattenrike remained uncoordinated, and despite its Ramsar designation, inventories and observations indicated that the natural and cultural values of the wetlands continued to be degraded. In particular, there was evidence of declining bird populations, eutrophication and overgrowth of the lakes, and a decrease in the use of flooded meadows for haymaking and grazing (Olsson et al. 2004).

In 1989, growing concern about the plight of the wetlands provided the impetus for establishing the Ecomuseum Kristianstads Vattenrike (EKV), largely through the efforts of a
key visionary individual, Sven-Erik Magnusson (Olsson et al. 2004). The EKV acts as a bridging organization to facilitate and coordinate wetland management activities (Hahn et al. 2006; Olsson et al. 2007). The EKV brought about a transformation in the governance of the wetlands in three key respects: i) It integrated conservation with economic and social development priorities, and led to the development of a new identity for the town of Kristianstad (which was previously known as a military training center), ii) It provided a framework for linking previously fragmented and uncoordinated individual efforts to manage the wetlands, and iii) It initiated a process of collaborative governance, creating a space for developing a common vision for the wetlands and addressing potential conflicts in a proactive manner. Institutionally, the EKV is part of the Kristianstad Municipality and reports directly to the municipality board, but has no authority to make or enforce legal rules (Olsson et al. 2004).

In June 2005 the Kristianstads Vattenrike became formally designated as a Biosphere Reserve under the Man and Biosphere Program of the United Nations Educational, Scientific and Cultural Organization (UNESCO). The EKV subsequently became known as the Kristianstads Vattenrike Biosphere Office, and continues to play a highly active and influential role in managing the Kristianstads Vattenrike.

Sabie River, South Africa

The Sabie River is a perennial river in eastern South Africa, regarded as one of the country's flagship rivers due to its high aquatic diversity and good ecological condition (Venter and Deacon 1995; CSIR 2001; O'Keeffe and Rogers 2003). The upper catchment (watershed) is
intensively used for commercial timber plantations, while the central subtropical region supports commercial and subsistence irrigation. The central catchment is also heavily relied upon for domestic water supply by impoverished rural communities. The lower catchment falls within the Kruger National Park and associated private game reserves, which constitute a premier tourist destination in South Africa (CSIR 2001). Upstream extractive uses of the river therefore compete directly with downstream subsistence, tourism and non-extractive conservation needs.

In March 1992, increased withdrawals combined with a major drought led to some of the lowest flows ever recorded in the Sabie River. It was projected that the river would run completely dry by July/August 1992 for the first time in recorded history (Venter and Deacon 1995). Given the high biodiversity value of the river, this situation was of significant concern to aquatic scientists in the Kruger Park. The situation was also of substantial concern to irrigation farmers and domestic water users. Prompted by their concerns, aquatic scientists in the Kruger Park called a meeting with the Sabie River Irrigation Board. Discussions resulted in the formation of a task team to investigate options for addressing concerns surrounding the drought, and subsequent stakeholder meetings led to the formation of the Sabie River Working Group (SRWG).

The SRWG acted as a bridging organization, transforming the existing governance system by i) Promoting dialog between the major stakeholders in the catchment, ii) Providing a forum where a common vision and goals for the river were developed and agreed upon, and iii) Providing an impetus for developing and implementing activities to realize this vision. The
river management objectives developed by the SRWG focused on the fair distribution of water within the catchment and preventing the river from drying up or becoming polluted within the Kruger Park. The different stakeholder groups proposed and carried out activities to realize these goals. These measures enabled the SRWG to meet its objectives, and the river never stopped flowing during the drought (O’Keeffe and Rogers 2003).

The SRWG remained active as a coordinator and facilitator of activities in the catchment for about a decade after its formation. The group was eventually disbanded during attempts to establish the Inkomati Catchment Management Agency (CMA), a government-initiated agency mandated by new environmental legislation that is to fulfill similar functions to the SRWG. However, the Inkomati CMA has yet to become operational, and the disbanding of the SRWG has created an important vacuum in the coordination of catchment activities.

**Yahara Lakes, Wisconsin, USA**

The Yahara Lakes — Mendota, Monona, Waubesa and Kegonsa — and their catchments lie almost entirely in Dane County, Wisconsin (Carpenter et al. 2006). Much of the watershed is urbanized, and the shorelines of Lakes Mendota and Monona are distinctive features of downtown Madison, the state capital. The lakes are heavily used for water-based recreation and have high aesthetic value. With the University of Wisconsin situated on the shores of Lake Mendota, they are also among the most studied lakes in the world (Brock 1985; Kitchell 1992; Carpenter 2003). The remainder of the Yahara watershed is mostly used for agriculture. Phosphorous runoff, mainly from urban construction sites and agricultural fields, is a major contemporary cause of algal blooms in the lakes, which lead to fish kills, affect
recreational activities, produce a nasty smell, and increase the costs of water purification (Carpenter et al. 2006). Valued ecosystem services are therefore compromised by the side-effects of agriculture and expanding urbanization.

Noxious algal blooms and fish kills have been regularly reported in the Yahara Lakes since the 1880s (Brock 1985). Initially the main cause of the eutrophication was runoff from cleared agricultural lands (the area was settled in the 1830s) and raw sewage disposal. Initiatives to divert sewage were active from the 1940s through the 1960s, and all sewage was diverted to sewage treatment plants by the early 1970s (Carpenter et al. 2006). However, nonpoint phosphorous pollution from fertilizer runoff and urban construction sites has led to ongoing eutrophication problems. Beginning in 1975, the Dane County Regional Planning Commission began preparing plans to control nonpoint pollution, but action was limited (Carpenter and Lathrop 1999). In 1980, the western half of Lake Mendota's drainage basin (Sixmile-Pheasant Branch Creeks) was designated as a priority watershed project by the Wisconsin Department of Natural Resources, allowing state funds to be used to share the cost of nonpoint pollution reductions with individual landowners and municipalities. However, voluntary participation by landowners was minimal, and many of the management practices turned out to be ineffective at reducing phosphorous inputs (Carpenter and Lathrop 1999).

To more effectively address the ongoing water quality problems in the Yahara Lakes, the Dane County Board created the Dane County Lakes and Watershed Commission (LWC) in 1988. The LWC was intended to integrate fragmented watershed management activities, and
was empowered to improve water quality and the scenic, economic, recreational and environmental value of the county's water resources (Carpenter et al. 2006). The LWC can recommend programs, plans and projects to the County Board for approval, and that minimum standards be passed for water quality benefits. The establishment of the LWC represented a transformation in the management of the Yahara Lakes in the following respects: i) It helped refocus efforts to manage water quality to account for activities in the entire watershed, rather than only focusing on in-lake management, ii) It provided a means of integrating fragmented efforts relating to the management of the Yahara Lakes and their watersheds, and iii) It allowed for formal representation of stakeholders and provided forums where the public could give input on lake management issues (Born and Rumery 1989; Nakamura and Born 1993). Although public participation had been emphasized prior to the LWC, the myriad governmental entities previously responsible for water-related management made effective public participation in decision-making processes difficult (Nakamura and Born 1993).

As of present (2008), the LWC continues to play an active role in coordinating freshwater governance in the Yahara Lakes district. There are, however, concerns that the LWC is preoccupied with dealing with short-term crises (e.g., restricting boat speeds to prevent flooding of shoreline houses due to high water levels), rather than focusing on long-term strategic issues, such as the risk of alien invasives and the potential consequences of climate change.
Factors underlying innovation in the case studies

Based on the literature and interviews, we identified important factors contributing to innovation and transformation in each of the case studies. We grouped these factors along three dimensions: i) The trigger or impetus for innovation, ii) Bricolage and the sources of new ideas and approaches, and iii) Diffusion, whereby new ideas and approaches became adopted and implemented (Table 2). As evident from the case studies, these dimensions seldom occur in sequence. Rather, they tend to occur concurrently with multiple iterations between the different components (Born & Genskow 2000; Wondolleck & Yaffee 2000; The Young Foundation 2006).

We used the analysis in Table 2 to explore the commonalities and differences between the studies, and suggest key common factors that facilitated innovation in ecosystem governance across the different case studies. These factors include environmental crises, reframing of perspectives, engaging stakeholders, leadership, and institutional support. These findings corroborate existing work in diverse fields. What our analysis adds is to emphasize specifically how these factors may influence social innovation and transformation in ecosystem governance, by enabling the development, adoption and implementation of new ecosystem governance approaches in modern Western societies.

Environmental crises

Ongoing environmental degradation, coupled with further anticipated degradation in future, appears to have been the major impetus for developing and adopting new approaches to ecosystem governance in all three studies (e.g., Table 3, Quote A). This observation confirms
work by other authors who suggest that ecosystem governance changes are often triggered by environmental crises (Plummer and Fitzgibbon 2004; Folke et al. 2005). However, in the case studies we examined, degradation itself appeared insufficient to precipitate changes in the governance approach, as least initially. It seems that at least two additional factors were necessary before alternative approaches to ecosystem governance were adopted.

First, the value of the ecological attributes being lost and their connections to human activities had to become appreciated. In all three studies, this growth in awareness seems to have been a gradual process involving a complex interplay between increased ecological understanding and changing societal values, and is further explored under the next point, *Reframing perspectives*. Second, transformation did not occur until it became apparent that existing approaches, or incremental modification to existing approaches, did not adequately address the degradation. For instance, the initial response to wetland degradation around Kristianstad was to protect the wetlands through designation as a Ramsar Wetland site, a response that fitted the sectoral approaches of the day. It then took more than a decade to “test” this intervention, realize its inadequacies, and initiate an alternative response in the form of the EKV (Olsson et al. 2004). In the Yahara Lakes case, incremental responses to environmental issues led to a large proliferation of agencies and programs responsible for water-related issues, but with no mechanism for coordination amongst these agencies (Table 3, Quote B). Only once it became clear that this situation was untenable and limited effective management of the lakes were alternative approaches sought (Born and Rumery 1989; Nakamura and Born 1993).
Nevertheless, all three case studies displayed a substantial degree of proactiveness in identifying and addressing environmental situations that would likely have led to considerable conflict and acrimony if they had been left unaddressed. Encouragingly, this suggests that actors are able to initiate and mobilize governance changes based on information that serious problems may be looming, without experiencing such problems first-hand. Based on the interviews we conducted, we speculate that ecosystem governance transformations may be more likely, and are more likely to be lasting, if initiated before high levels of conflict set in. It is well-known that once strong feelings of acrimony exist, people tend to become locked into defensive positions (Kahane 2004; Tavris & Aronson 2007). Initiating dialog and developing and implementing new approaches to ecosystem governance may then become substantially more difficult (Homer-Dixon 1995; Kofinas and Griggs 1996; Olsson 2007).

Reframing perspectives

In the case studies we examined, the reframing of perspectives in several domains appeared central to the emergence of approaches that transformed ecosystem governance. In all three cases, reframing of perspectives seems to have been substantially informed by new ecological understanding. For instance, it was not until the 1960s that the link between phosphorus in agricultural fertilizers and freshwater eutrophication became firmly established scientifically (Rohlich 1969; Schindler 1974). In the case of the Yahara Lakes, understanding the connections between activities in the watershed and in-lake water quality problems was central to identifying the need for management at the watershed rather than simply at the
lake level. The newly recognized need for watershed level management was central to the development of idea for the LWC and the drive to establish it (Born and Rumery 1989).

Reframing of perspectives in the case studies also appears to have been greatly influenced by more diffuse processes of growing environmental awareness and changing societal values. Increasing environmental awareness globally and associated changes in societal values through the 1960s and 1970s (e.g., Carson 1962; Ehrlich 1968; Meadows et al. 1972) appears to have made stakeholders in the different studies more receptive to new governance initiatives focused on improving environmental quality. For instance, in the case of Kristianstad, media attention highlighting the death of seals along the Swedish coast in the late 1980s was specifically noted as helping create a receptive environment for discussions on establishing the EKV (Olsson 2007).

More directly, stakeholder perspectives seem to have been influenced by exposure to the local ecosystems and informal interaction with other stakeholder groups. For instance, interviewees in the Sabie River case study identified field trips as a critical success factor in the development of the SRWG. During these field trips, irrigation farmers, forestry representatives and other stakeholders were taken out to a section of the Sabie River in the Kruger Park where they electrofished and collected macro-invertebrates. Interesting and unique aspects of the ecology were highlighted in an informal and enjoyable way. Similar outings were arranged by the forestry and irrigation sectors to highlight the ways in which they used and cared for the river system. These outings were seen as critical to building appreciation and understanding of the different stakeholders' needs, and fostering an
attachment to the river ecosystem. Similar field trips and the establishment of a river boat
cruise in the Kristianstad wetlands were noted as having played an analogous role in
reframing perspectives and shifting values in that study (e.g., Table 3, Quote C).

At a more fundamental level, reframing the idea of environmental conservation itself seems
to have been a critical factor enabling the emergence of alternative governance approaches,
especially in the Kristianstad case. Prior to the establishment of the EKV, conservation in
the Kristianstad region was largely seen as involving the protection of the wetlands through
the exclusion of human influence. A crucial factor in convincing the Kristianstad
Municipality to establish the EKV was that it proposed to link conservation to economic
and social development priorities. The wetlands were reframed as being an asset rather than
a problem, so that the region became seen as "water rich" as opposed to "water sick"
(Olsson et al. 2004). The stated vision of the EKV was to open the wetlands up for human
use and enjoyment rather than excluding people (Table 3, Quote D). Reframing
conservation in a way that integrated economic and social imperatives and recognized and
incorporated human use of ecosystems was also a distinguishing feature in the establishment
of the LWC and the SRWG.

Lastly, the Kristianstad case study suggests that role models of integrated, collaborative
ecosystem management can be important as a source of inspiration for developing
alternative ecosystem governance approaches. Magnusson was much inspired by the French
Musée Camarguais and the UNESCO Man and Biosphere Program to develop the original
proposal for the EKV (Olsson et al. 2004).
**Engaging stakeholders**

In all three case studies, the engagement of key stakeholders appears to have been a central element in developing new approaches to ecosystem governance and enabling these ideas to be successfully implemented. Stakeholders included key groups impacting or impacted by changes in the local ecosystem, as well as various fragmentary groups carrying out conservation and management efforts. A simple, compelling focus seems to be central to initially engaging stakeholders and enabling a collaborative group to form and “gel”. For example, in the Sabie River case, preventing the river from running dry provided a clear issue for approaching different stakeholders and initiating discussions. Once the group had formed, it was possible to develop more complicated and diverse foci. Similarly, discussions leading up to the formation of the LWC focused primarily on water quality management (Born and Rumery 1989), but over time the LWC adopted additional foci such as control of non-native species.

Several interviewees noted specifically that successfully engaging stakeholders does not simply involve inviting everyone who may be interested in a particular issue to a meeting (e.g., Table 3, Quote E). Rather, different stakeholder groups are initially best approached individually, so that discussions can be framed in terms that speak to the concerns and needs of the different groups. The development of the EKV provides a very clear example of this approach. After coming up with the initial idea for the EKV, Magnusson garnered support by approaching receptive, strong individuals in key organizations (Table 3, Quote F). Over time close relationships and trust with these individuals was established, and only once Magnusson had incorporated their ideas into the EKV proposal and had their support did
he involve a broader spectrum of stakeholders (Olsson et al. 2004; Olsson 2007). This “starting small” approach remains a key strategy of the EKV when starting new projects. It also supports the well established finding that innovative new ideas need initial nurturing if they are to grow (The Young Foundation 2006; McKeown 2008).

Once collaborative groups in the different case studies had formed and agreed on a set of objectives, the experiences and perspectives that different stakeholders brought to the table played an important role in developing and implementing strategies for meeting the groups’ objectives. For instance, to achieve the agreed-upon goal of preventing the Sabie River from running dry, the forestry sector in the region volunteered to remove “runaway” exotic trees in the riparian zone of the river’s upper reaches, while irrigators voluntarily developed and implemented water restrictions (which were subsequently adopted and promulgated by the Department of Water Affairs and Forestry) (O’Keeffe and Rogers 2003). Interviewees in the Sabie River case felt that the development of a “team spirit” was central to motivating the different actors to generate and voluntarily implement such ideas. The field trips mentioned in the previous section, which also involved barbequing and drinking beer, were believed to be central in building the trust and respect that enabled a team spirit to emerge (Table 3, Quote G). As found by other authors (Born & Genskow 2001), discussing and developing organizational rules for the functioning of the group also appeared to play an important role in the development of group cohesion in the different case studies.

In the LWC, the possibility for engaging key stakeholders has arguably been partly compromised by over-formalization of the processes by which the group operates. Rather
than constituting a voluntary association of interested and concerned groups and individuals, the LWC constitutes a formal commission to which members are appointed. Membership is limited to ten people that represent specific rural and urban constituencies. While this may ensure that the views of certain groups are not excluded, it makes the LWC somewhat inflexible with regard to engaging new groups or individuals that become relevant or interested to contribute to the work of the LWC. An over-formalized structure may also compromise learning and innovation, as many authors have found that new ideas and collaboration are best generated in trustful, informal settings (Westley and Vredenburg 1997; Hahn et al. 2006; McKeown 2008).

**Leadership**

Leadership was identified as a critical element in the development of new governance approaches in all three case studies, supporting a large body of existing work on the importance of leadership in transformation (Born & Genskow 2001; Folke et al. 2005; Olsson et al. 2006; Westley et al. 2006). In the studies we examined, leadership played a particularly important role in three respects: i) Reframing perspectives, especially by providing or facilitating the development of an alternative vision for ecosystem management, ii) Engaging key stakeholders, fostering a group identity, and building networks, and iii) Managing conflict. These functions were often performed by the same individual, but in some cases were dispersed across several individuals.

Since (i) and (ii) have largely been covered in the above discussion, we focus here on conflict resolution. Tensions between the perspectives and needs of different stakeholders are often
a key source of new ideas and approaches. For instance, the need to integrate conservation with economic and social development needs was central to the development of the EKV approach. However, if not appropriately managed, conflicts can forestall the development and spread of new approaches and ideas (Kofinas and Griggs 1996; Olsson 2007). In the Sabie River, for example, there was substantial antagonism between the forestry and irrigation sectors during the initial meetings (Table 3, Quote H). Despite established evidence that exotic timber plantations substantially reduce runoff (Wicht 1949; Hewlett and Hibbert 1961; Smith and Scott 1992), representatives of the forestry sector maintained that the plantations in fact improved runoff. The chairman of the group, Japie Lubbe, is credited with managing the impasse and enabling agreement to be reached on the major factors impacting river flow. Only once consensus was attained in this respect could options for reducing impacts be productively explored.

In terms of building networks and diffusing ideas, we add that leaders in the different case studies acted as key nodes linking a variety of local networks. Individuals that performed key leadership functions were typically involved in leadership roles in several other networks. For instance, Shary Bisgard, an initiator and early chair of the LWC, was also actively involved in several other citizen and county groups, such as neighborhood associations and county planning boards. At any one time, as well as through their movement between groups over time, such individuals create links between groups and function as a vehicle for diffusion of ideas from one group to another (Janssen et al. 2006; Westley et al. 2006).
Institutional support

Institutional support, specifically governmental support, appeared to be a key factor enabling the bridging organizations in the different case studies to become established. In the case of the EKV and the LWC the organizations are formally housed by the local government, who provide salaries and office space for a small core staff. Such assured funding, especially for managing administrative and organizational aspects of bridging organizations has been found to be central to their maintenance over time (Born & Genskow 2001; The Young Foundation 2006). Links to and support from government also provide a potential mechanism for diffusion of ideas and activities of the bridging organizations to other areas and groups (The Young Foundation 2006).

The Sabie River case illustrates, however, that government can also inadvertently squash grassroots-initiated bridging organizations. Initially, the Department of Water Affairs and Forestry was very supportive of the activities of the SRWG. However, new legislation introduced in 1998 (Republic of South Africa 1998) mandated the establishment of Catchment Management Agencies (CMAs) for all major catchments in South Africa. These agencies are to contain representatives from all major stakeholder groups, and are empowered to make decisions about local water allocation and to perform various bridging functions in managing local river systems. The legislation is widely heralded as visionary (Postel & Richter 2003; Gowlland-Gualtieri 2007) and is based on contemporary understanding of complex systems and values espousing stakeholder participation and empowerment. However, it is widely held that the organizational culture of the Department is poorly matched to facilitating the establishment of these agencies (Rogers et al. 2000), and
in addition the Department is faced with serious capacity constraints. Attempts to establish the Inkomati CMA for the Sabie River led to the disbanding of the SRWG, and the subsequent faltering of the process is a source of much frustration.

Implications: what strategies might foster a wave of innovation in ecosystem governance?

In many respects society at the global scale can be interpreted as poised for, or perhaps already undergoing, a major back loop (Gray 1995; Steffen et al. 2004; Constanza et al. 2007). Environmentally, many of the approaches and practices that characterized the modern, industrial era of the last 300 years or so are now widely regarded as unsustainable and in need of fundamental reorganization (Steffen et al. 2004; MA 2005; IPCC 2007). Furthermore, although often overlooked, dramatic changes in attitudes toward the environment, support for inclusive, democratic practices, and beliefs about the ability and role of science in society have occurred over the past half century (Funtowicz and Ravetz 1993; Gray 1995; Cortner & Moote 1999; Ziman 2000). Our case studies suggest that these conditions create fertile ground for radical social innovation. The question we ask here is how might such innovation be fostered and enhanced?

Applying a social innovation framework to understand transformation in ecosystem governance emphasizes that the creation of environments conducive to the development of alternative ideas for ecosystem governance, as well as conditions that nurture such ideas and allow them to take root, are critical to enabling transformation. Based on the case studies we examined, we suggest that the following strategies may help create environments which
enable new ideas for ecosystem governance to emerge and flourish, at least at the local scale. As noted previously, the exploratory nature of this study means that the effectiveness of these strategies, and potential ways of enhancing the strategies need to further examined in more detailed research studies.

**Foster environmental awareness and attachment to local ecosystems**

Our interviews highlighted that leaders who spearhead transformations are often greatly motivated by awareness and understanding of the unique aspects of the ecosystems they seek to have better governed, and often feel a deep attachment to these ecosystems. In many cases, such attachment started forming during the leaders’ youths. Additionally, our case studies indicate that stakeholders become substantially more open to changes in ecosystem governance, often at a cost to themselves, once they are aware of the value of the local ecosystems and have personally experienced these environments. We therefore suggest that fostering awareness and attachment to local ecosystems, among both adults and the youth, may directly and indirectly contribute to increased rates of social innovation in ecosystem governance. This supports work by other authors (Vredenburg and Westley 1997). Our case studies suggest that environmental awareness and attachment to this end may be best fostered through informal, experiential activities that take place within the setting of the local ecosystem (e.g. Table 3, Quote G). Our interviews also point to the importance of access to local ecosystems in building support for their conservation (e.g., Table 3, Quote D). Although many environmental awareness activities already exist, it is clear that there is substantial scope for expanding modern-day interaction with local ecosystems (Louv 2008).
Expansion would be greatly aided by increased funding availability for initiatives that aim to improve access to and awareness of local ecosystems.

**Foster environmental leadership**

As in our case studies, leadership has widely been found to be critical to the transformation process (Born & Genskow 2001; Olsson et al. 2004). The nature, effectiveness and development of leadership is an area that has received much attention, especially in the business literature (Lewin et al. 1939; Greenleaf 1977; Burns 1978; McCauley & Van Velsor 2003; House and Podsakoff 2003). Studies have shown that personality traits strongly influence an individual’s leadership ability (McCauley & Van Velsor 2003), although most people can develop their leadership effectiveness through focus, practice and persistence (McCauley & Van Velsor 2003; Carter et al. 2004) and many leadership development programs exist to meet this need. However, relatively few programs focus specifically on leadership development for addressing environmental issues. Leadership skills for promoting integrated, collaborative adaptive environmental governance require what have been termed democratic or servant leadership, where all parties contribute to and take ownership of key decisions (Lewin et al. 1939; Greenleaf 1977). Leadership for environmental issues also requires social entrepreneurship: the ability to recognize a social problem and use entrepreneurial principles to organize, create, and manage a venture to address the problem (Leadbeater 1997; Bornstein 2004). Key strategies employed by successful social entrepreneurs include i) building and amplifying networks of individuals and organizations relevant to the problem, ii) dispersing power, for instance by involving people such as technicians who have low formal status but hold knowledge critical to the problem, and iii)
avoiding centralized control and structuring (Westley and Vredenburg 1997; Westley 1997). We suggest that developing and expanding programs that focus specifically on leadership development for collaborative problem solving around environmental issues could give a substantial boost to social innovation in the environmental governance arena.

Leadership could also potentially be leveraged by identifying and supporting key, established, innovative individuals already performing leadership functions in support of environmental governance transformations in a particular region. As evident from our case studies, such individuals have well-established, extensive networks, and act as key links between disparate groups and initiatives. Providing key individuals with financial and institutional support, and giving them relative carte blanche to initiate and carry out activities is increasingly adopted as a strategy for enhancing innovation (The Young Foundation 2006; Westley et al. 2006; McKeown 2008).

**Foster dialogue between key stakeholders**

Our case studies suggest that dialogue between key stakeholders is a critical precursor to the establishment of integrated, collaborative ecosystem governance institutions. Our case studies also indicate that simply inviting all stakeholders to a few joint meetings is usually insufficient for achieving the generative dialogue that enables new ideas to emerge and grow (e.g., Table 3, Quote E). Much recent work can be drawn upon to promote more productive dialogue between parties with conflicting interests (Gray 1989; Isaacs 1999; Kahane 2004; Senge et al. 2004). This work stresses the need to first acknowledge and explore the perspectives and needs of the different parties. Then, rather than seeking to negotiate a
compromise between the initially articulated needs of the different parties, the emphasis is on reframing the situation away from one of conflict to one of common interests and challenges (Wondolleck & Yaffee 2000; Rogers 2006). Such reframing often enables novel and durable strategies to emerge that could not be imagined if the situation is framed in terms of conflicting interests. Although dialogue processes demand substantial time and commitment by participants and may require management by experienced facilitators, our work suggests that the understanding, trust, and networks built through dialogue processes probably justify greater investment in such activities as a means of stimulating transformation in ecosystem governance. Alternatively, as shown in our case studies, dialogue processes may be realized by supporting key leaders who can lead and support one-on-one meetings between stakeholders in a particular region (e.g., Table 3, Quote F).

Although not employed in the case studies we examined, we also note that scenario planning may be a particularly productive and non-threatening means by which dialogue processes can be realized (van der Heijden 1996; Peterson et al. 2003; Kahane 2004; Scearce et al. 2004). Scenario planning has two additional benefits: i) It explicitly requires that several alternative futures for a region be considered, which encourages broader framing and scoping of a situation and hence stimulates the generation of a more diverse set of possible governance options, and ii) By focusing on the future, it removes participants from immediate conflicts, which is often a significant obstacle to initiating dialogue (Kofinas and Griggs 1996; Olsson 2007). Exploring the use of scenario planning as a tool in ecosystem governance transformation may be a particularly fruitful area of research.
Provide institutional support

Our case studies point to the importance of institutional support in enabling new ecosystem governance approaches to take root. As noted in the preceding points, supporting environmental awareness, leadership development and stakeholder dialogue processes can stimulate the formation of groups aimed at collaborative ecosystem management. However, once such groups have formed their maintenance over time is often seriously challenged by institutional and financial constraints (Wondolleck & Yaffee 2000; Born & Genskow 2001). As noted by other authors (Born & Genskow 2001; The Young Foundation 2006), the Kristianstads Vattenrike and Yahara Lakes examples suggest that partnering with local government institutions often provides a durable base from which collaborative ecosystem governance groups can function. However, in many parts of the world, enabling such partnerships requires major changes in how governments operate. In particular it requires strategies and mechanisms that prioritize creative connections and institutions that cut across traditional boundaries (The Young Foundation 2006). Furthermore, given the complex and multi-dimensional nature of the issues that collaborative governance groups seek to address, the outcomes from the activities of these groups may not be obvious or may only become evident after a substantial period of time. Born and Genskow (2001) therefore stress the need for more complex evaluation procedures to assess the performance of such groups. Specifically, assessment should not focus only on environmental outcomes, but should also include measures of social capacity development, institutional changes, economic outputs and intermediate environmental outcomes (Born & Genskow 2001). Our case studies further emphasize the importance of fun, informal activities, such as field trips and barbequing, in fostering team spirit and innovation (e.g., Table 3, Quote G). Such activities challenge
conventional notions of appropriate activities for environmental management groups and agencies. Our findings therefore highlight the need to potentially rethink restrictions associated with funding that aims to support and stimulate innovation in ecosystem governance.

We also note that several countries have piloted innovative legislative and institutional arrangements to stimulate the formation of collaborative grassroots ecosystem management groups. For instance, Sweden passed legislation enabling the formation of local fishing associations in the early 1980s. These associations have access to governmental funding, as well as certain legal powers relating to the setting of fishing quotas, and management of land use in the watersheds of the lakes which they manage (Olsson and Folke 2001). At least in some regions, there is evidence that the formation of such associations have stimulated innovative ecosystem management strategies and have improved environmental conditions (Yaffee et al. 1996; Born & Genskow 2000; e.g., Olsson and Folke 2001). Other government-mediated factors that can strongly influence innovation in ecosystem governance are the creation of markets for ecosystem services (Dasgupta 2001; Daily & Ellison 2002), the provision of capital, education and infrastructure (Homer-Dixon 1995), and political stability (Wright 1991; Homer-Dixon 1995). Central to governmental strategies for stimulating innovation in ecosystem management is an acknowledgement that not all funded initiatives will succeed. Like in any innovation process, intolerance of failure is likely to stifle innovation in ecosystem governance.
Conclusion

The findings from this exploratory study suggest that a social innovation perspective provides a useful alternative framework for studying and understanding factors that may promote transformation in ecosystem governance. Our insights complement the work of Olsson et al. (2004; 2006) on the phases of transformation (Fig. 1). Specifically, our analyses suggest that the first phase of transformation identified by Olsson et al. (2004), “Preparing for change”, largely involves i) the creation of environments that enable ideas for alternative approaches to ecosystem governance to be generated and explored, and ii) developing the networks, trust and other conditions that enable new ideas to be adopted when a suitable “window of opportunity” arises (Kingdon 1995; Olsson et al. 2004; Olsson et al. 2006; Westley et al. 2006). In the preceding section we have suggested several strategies by which these two factors may potentially be enhanced in support of ecosystem governance transformation. However, our findings are exploratory in nature and need to be verified and further explored by more in-depth studies. Our insights would also be furthered by in-depth analyses of ecosystem governance transformations at larger regional, national, and international scales, and with respect to environmental attributes other than freshwater.

We note two final issues in relation to social innovation and transformation in ecosystem governance. First, innovation is a strongly nonlinear process. Transformative social innovations are often characterized by a protracted period in which new ideas are around but have very limited rates of adoption. However, as in technology adoption, once uptake of social innovations reach a certain level, they may suddenly “take off” so that adoption rates increase dramatically (Rogers 1995). Our analyses suggest that in terms of meeting new social
and environmental needs triggered by environmental degradation, two important reasons for lags in ecosystem governance transformation are i) that it takes time to appreciate the value of the ecological attributes being lost and to establish their likely causes, and ii) that initial responses are typically incremental changes to existing approaches, and only once it is established that such responses are insufficient are more radical and innovative responses sought. Together, these two factors suggest that we should often expect a substantial delay between the time at which environmental degradation becomes apparent, and the time at which society reorganizes ecosystem governance to address the degradation. One interpretation of the current global environmental situation is that it is precisely in this “lag” phase: environmental degradation is apparent, the value of the ecological attributes being lost is increasingly appreciated, it is recognized that existing approaches are inadequate (MA 2005; IPCC 2007), but social institutions have not yet reorganized to address the situation. Our objective has been to highlight factors that might help trigger and foster such reorganization.

Second, innovation is always defined relative to a particular context and time. What is new and innovative today is set to become old and the source of new problems in future. We therefore do not espouse integrated, collaborative governance as a panacea. Although we concur with authors who maintain that there is enormous scope to expand such forms of governance, and that they are likely to help address many important environmental challenges we face today (Cortner & Moote 1999; Wondolleck & Yaffee 2000; Armitage et al. 2007), integrated, collaborative governance is not appropriate in all contexts and is certain to generate its own set of problems in time. A critical challenge in ameliorating the emergence
of new problems is the design of ecosystem governance institutions that remain innovative and adaptive over time (Gunderson & Holling 2002; Berkes et al. 2003; Chapin et al. In press). We have not investigated this issue in our analyses, but identify it as an important area for further research. In particular, we note that there is tremendous scope for social innovation to meet ongoing needs for adaptation as ecosystem governance evolves in different localities. Creating environments that foster ongoing social innovation are likely to be critical in this respect, and may be usefully informed by understanding gained in the business and science arenas.

Acknowledgements

We greatly thank the interviewees for sharing their insights and experiences: Per Olsson, Sven-Erik Magnusson, Freek Venter, Andrew Deacon, John Magnuson, and Steve Born. Amy Kamarainen is thanked for comments on the draft manuscript. Our research time was funded through a Fulbright grant, and the North Temperate Lakes LTER program.

References


Table 1. Key literature sources used in each of the case studies.

<table>
<thead>
<tr>
<th>Kristianstads Vattenrike</th>
<th>Sabie River</th>
<th>Yahara Lakes</th>
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<tbody>
<tr>
<td>Schultz et al. (2007)</td>
<td>Unpublished Box drafted by Freek Venter on the origins of the Sabie River Working Group</td>
<td>Lakes and Watershed Commission website</td>
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</tbody>
</table>
Table 2. Key factors that triggered innovation in ecosystem governance, acted as sources of ideas for alternative governance approaches, and facilitated diffusion of new approaches in each of the case studies.

<table>
<thead>
<tr>
<th></th>
<th>Kristianstads Vattenrike</th>
<th>Sabie River</th>
<th>Yahara Lakes</th>
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</thead>
<tbody>
<tr>
<td><strong>Impetus for innovation</strong></td>
<td>Ongoing degradation of the wetland environment with declining bird populations, eutrophication and loss of flooded meadows.</td>
<td>Major drought which was projected to lead to the first-ever cessation of flow in the Sabie River.</td>
<td>Ongoing eutrophication problems related to phosphorous runoff from agricultural fields and urban construction sites.</td>
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<td></td>
<td>Increasing environmental awareness amongst the public, partly associated with the death of seals along the coast.</td>
<td>Ongoing increases in withdrawals, threatening future river flow.</td>
<td>Fragmentation of management activities and relating to lake management.</td>
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<td></td>
<td>Need to revive the town’s identity and local economy following its decline as a military training center.</td>
<td>Increasing awareness of the value of the Sabie River in terms of its aquatic diversity and relatively unimpacted state.</td>
<td></td>
</tr>
<tr>
<td><strong>Sources of alternative ideas and approaches</strong></td>
<td>French Musée Camarguais and the UNESCO Man and Biosphere Reserves provided inspiration for alternative, more integrated, governance approaches.</td>
<td>Growing ecological understanding (especially associated with the Kruger Park Rivers Research Programme) increased appreciation of the need for systemic approaches to river conservation.</td>
<td>Scientific expertise associated with the University of Wisconsin and the Wisconsin Department of Natural Resources.</td>
</tr>
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<td></td>
<td>Failure of existing approaches, such as Ramsar Wetland designation, highlighted the need for more integrated approaches.</td>
<td>The sector-specific knowledge of the different stakeholders enabled identification of actions to improve river flow without jeopardizing livelihoods – e.g., ringbarking riparian invasives, allocation of irrigation withdrawals.</td>
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<td></td>
<td>The knowledge and experience of different groups and agencies active in wetland conservation.</td>
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<tr>
<td><strong>Adoption and diffusion of new ideas</strong></td>
<td>Visionary leadership was critical to developing the new approach, gaining support for the approach, and providing the ongoing drive to get it implemented.</td>
<td>Leadership was central to engaging stakeholders and managing conflict between stakeholders.</td>
<td>Committed leadership and political know-how was central to developing a politically acceptable new governance approach.</td>
</tr>
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<td></td>
<td>Engaging stakeholders and gaining their support – largely through one-on-one</td>
<td>Field trips and social activities (e.g. barbequing) were central to building understanding of different stakeholder needs and a</td>
<td>Provision for representation on the new commission by key interest groups was central to making it</td>
</tr>
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meetings where the new approach was framed in ways that the different stakeholders could relate to — was critical to the adoption of the new approach.

Institutional support from the Municipality ensured continuity and support for core staff.

Institutional support from Dane County has enabled continuity of the new commission and support of core staff.

Granting of advisory powers to the commission strengthened it.

Team spirit of shared commitment to goals.

Developing commonly agreed upon objectives and goals for river management, supported by all key stakeholders, was central to adopting and implementing new governance approaches.
Table 3. Illustrative quotes highlighting factors facilitating transformation in the studies.

<table>
<thead>
<tr>
<th>Factors</th>
<th>Quotes</th>
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</thead>
<tbody>
<tr>
<td>Environmental crises</td>
<td>[Quote A] “Projections showed that the river would stop flowing completely by July or August of that year if no correctional actions were taken. This was almost incomprehensible as the Sabie River was well known for its high biodiversity (at that stage it was hailed as the most biodiverse river in South Africa) that was dependent on flowing water as habitat and was seen as a flagship of South African rivers.” (Sabie River)</td>
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<tr>
<td>Fragmentation</td>
<td>[Quote B] “Prior to 1988, the DNR [Department of Natural Resources] and RPC [Dane County Regional Planning Commission] were primarily responsible for coordinating water quality management through a county water quality plan... Watershed management consisted of numerous separate programs and actions carried out over time by different entities. Also, the DNR was responsible for in-lake management while municipalities and the county were responsible for shorelines, surface waters, and runoff.” (Nakamura and Born 1993) (Yahara Lakes)</td>
</tr>
<tr>
<td>Reframing perspectives</td>
<td>[Quote C] “Previously I was almost afraid of the authorities, it felt so bureaucratic somehow. But thanks to this project I have learned a lot and I have a completely different view now. It’s more like we all sit in the same boat.” (Hahn et al. 2006) (Kristianstads Vattenrike)</td>
</tr>
<tr>
<td>Engaging stakeholders</td>
<td>[Quote D] “SEM [Sven-Erik Magnusson] presented the area in a different way than anyone had done before and I became aware of the values. Many considered the wetlands as a problem... SEM presented a nature conservancy plan that didn’t close the area but opened it up and made it accessible for the public.” (Olsson et al. 2004) (Kristianstads Vattenrike)</td>
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<td>[Quote E] “Having talked to so many people out in the district we realize that bombs might be dropped if we were to bring everybody together for a large meeting. I mean, you don’t gather people if you don’t think anything positive will come out of the meeting.” (Hahn et al. 2006) (Kristianstad Vattenrike)</td>
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<td></td>
<td>[Quote F] “The key was to avoid a one-size-fits-all proposal that would be so neutral that nobody would be interested. Instead I [Sven-Erik Magnusson] had to approach each person and identify what their specific needs and interests might be and emphasize the parts of the [EKV] project proposal that they could identify with and find of interest.” (Olsson et al. 2004) (Kristianstads Vattenripe)</td>
</tr>
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<td></td>
<td>[Quote G] “The esprit de corp that formed in the SRWG was partly achieved by organizing field days in the respective areas and hosted by the different sectors. Meetings were held biannually under the trees at the Kruger Gate gauging weir and delegates were treated with some beers, soft drinks and a braai [barbeque] afterwards.” (Sabie River)</td>
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<tr>
<td>Leadership</td>
<td>[Quote H] “Initially there was considerable conflict especially between the irrigation and forestry sectors, which accused each other of unsustainable practices. Outstanding leadership and fostering the notion that we are all together in the catchment and cannot wish each other away eventually led to a cohesive committee as rivalry made way for practical jokes. Friendships that were molded in those years still last up to this day.” (Sabie River)</td>
</tr>
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</table>
3. Building resilience of the transformed system

**Figure 1.** The adaptive cycle (Holling 2001; Gunderson & Holling 2002) provides a useful metaphor for understanding incremental and radical innovation in complex SES. The front loop of the adaptive cycle can be seen as largely characterized by incremental innovation, while the back loop is typically marked by radical innovation. Factors that trigger a switch from the front loop to the back loop often derive from processes operating at larger or smaller scales than that of the system of interest (Holling et al. 2002). “Traps” in the adaptive cycle may also be seen in the context of innovation. A “poverty trap” refers to a situation where the system is unable to move out of the back loop due to a lack of new ideas or an inability to choose an option and act upon it. A “rigidity trap” results from resistance to the adoption of new innovations due to, for instance, large rigid bureaucracies or powerful groups with vested interests (Miller 1993; Holling et al. 2002). Boxes indicate the phases of transformation in ecosystem governance identified by Olsson et al. (2004), which we have mapped onto the adaptive cycle.
Figure 2. Location and key statistics of the three case studies. The three case studies show substantial cultural and economic diversity. Sweden is a relatively homogeneous society, as is Wisconsin, whereas South Africa is highly multi-cultural. The United States has a strong capitalist and individualist ethic, whereas Sweden is a socialist society; South Africa lies somewhere between these extremes. In terms of economies, Sweden and the United States are both highly developed, while South Africa is a strong emerging economy (World Bank 2006). Population data are interpolated estimates for the watersheds for *circa* 2000. Photo credits: Dick Lathrop; Patrik Olofsson, Melissa Parsons.
CHAPTER 5

PREPARING FOR THE FUTURE:

TEACHING SCENARIO PLANNING AT THE GRADUATE LEVEL

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Abstract

Are environmental science students obtaining the mindsets and tools needed to help address the considerable challenges posed by the 21st century? Today’s major environmental issues are characterized by high-stakes decisions and high levels of uncertainty. While traditional scientific approaches are critical, contemporary challenges also require new tools and new ways of thinking. We provide an example of how such “post-normal” approaches have been taught at the graduate level through practical application of scenario planning. Students found the course highly stimulating, thought-provoking and inspiring. Key learning points were recognizing the need for multiple points of view in understanding complex environmental issues, and better appreciating the pervasiveness of uncertainty. Collaborating with non-academic stakeholders was found to be particularly insightful and enjoyable. Most students left the course feeling more positive and inspired about the potential contribution they can make to addressing the environmental challenges society faces.

In a nutshell

♦ Many pressing environmental issues require skills and approaches to problem-solving not traditionally taught at universities. These include acknowledging diverse values and worldviews, dealing with high levels of uncertainty, and working in collaborative interdisciplinary teams.

♦ Scenario planning provides a tool for teaching such skills at the graduate level, as a complement to traditional scientific training.

♦ Environmental science students found exposure to these approaches stimulating, refreshing and inspiring.
Introduction

Fast-growing human populations, wealth and technology are placing unprecedented pressures on the planet (Steffen et al. 2004; MA 2005c; IPCC 2007). Are tomorrow’s environmental scientists being appropriately prepared to help address the challenges this situation poses for society? Future environmental research will largely focus on established issues such as climate change and biodiversity loss. However, the context for this work is likely to differ increasingly from that in which science has traditionally been conducted and is still predominantly taught (Funtowicz and Ravetz 1993; Gibbons et al. 1994; Ziman 2000). Environmental issues are becoming ever more politicized because of the rising stakes they hold for society. Consequently, the way environmental challenges are defined, the methods used to study them, and the interpretation of results are increasingly scrutinized and contested. This scrutiny is also highlighting the extent to which many scientific endeavors are underpinned by values and assumptions that may favor certain sectors of society above others (Bocking 2004; Sarewitz 2004).

At the same time, it is becoming apparent that most environmental issues are characterized by substantial and often irreducible uncertainties (e.g., Pilkey & Pilkey-Jarvis 2007). This owes to the complex nature of environmental systems: they involve large numbers of interdependent components that span multiple spatial and temporal scales, and are self-organizing and constantly evolving (Holling 2001; Manson 2001). These characteristics create inherent limitations for the predictability of complex systems. A prime example is that despite substantial investment in research and tremendous growth in computing power, uncertainties in future climate change projections have not lessened appreciably over the
past 30 years (Roe and Baker 2007). Such intrinsic uncertainties are being exacerbated by rapid and novel changes in drivers of environmental change, such as population growth and new technologies.

High decision stakes and high levels of uncertainty set the stage for what has been termed “post-normal science” (Figure 1) (Funtowicz and Ravetz 1993). Post-normal science reframes the relationship between science and decision-making. The belief that scientists can provide certain, objective information on which to base policy is increasingly recognized as inappropriate for contemporary environmental challenges (Bocking 2004; Sarewitz 2004). In post-normal contexts far-reaching societal decisions have to be made, often with considerable urgency, on the basis of information that is riddled with uncertainties. In addition, facts and values often cannot be clearly separated. It is increasingly argued that the most effective approach under these conditions is a dialogue among all parties, where scientists take their place at the table together with other stakeholders. To ensure high quality decisions, the decision-making process itself and assessments of information quality become critical (Funtowicz and Ravetz 1993; Gibbons et al. 1994).

Post-normal science draws on traditional scientific tools but also demands new tools and, in particular, different ways of thinking (Bammer 2005; Norgaard and Baer 2005). A popular tool employed in post-normal scientific contexts is scenario planning (van der Heijden 1996; Peterson et al. 2003). Well-known examples of post-normal scientific processes are the Millennium Ecosystem Assessment (MA 2005c) and the Intergovernmental Panel on Climate Change Assessment (IPCC 2007). In post-normal science the aim is not primarily to
reduce or eliminate uncertainty but to identify and manage it. Differing values are made explicit and become part of the deliberations. The process is characterized by interactive dialogue between large interdisciplinary groups of scientists and other stakeholders, and the potential contribution of non-scientific knowledge is recognized. The outcomes of the process are evaluated by an extended peer community, which includes scientists as well as government officials, business representatives and citizens (Funtowicz and Ravetz 1993; Gibbons et al. 1994).

Despite the growing importance of these emerging tools and mindsets, graduate students, particularly in the natural sciences, typically gain little exposure to these approaches. Like traditional research skills, post-normal scientific skills are best acquired through practical application. The aim of this paper is to provide an example of how post-normal scientific thinking might be taught at the graduate level, using scenario planning. We emphasize that post-normal science does not replace the need for traditional scientific skills, but provides an important expansion of students' skill sets. We base our ideas on a seminar at the University of Wisconsin-Madison during which we worked with local community members to develop scenarios for the future of Lake Wingra in Madison, Wisconsin. The authors of this paper are the students of the class and our instructor. To qualify for authorship, students had to provide detailed written reflections on the course, draft a section of the paper, and review a full draft of the manuscript.
Scenario planning in brief

Scenario planning is a highly creative exercise that is particularly well-suited to considering complex systems, fundamental uncertainties and conflicting values (van der Heijden 1996; Kahane 2004). Scenario planning has been applied in a wide range of contexts, including political decision-making, business planning, and environmental management. The outcome of a scenario planning process is typically a set of 3-5 scenarios, or plausible stories about how the future might unfold. These qualitative storylines may subsequently be quantified, depending on available resources and the objectives of the process. Scenarios often describe unlikely or surprising futures. The intention is to provoke consideration of how critical uncertainties may affect the future, and thereby broaden perspectives, challenge assumptions, and highlight hidden dangers and opportunities. Scenario planning differs from other approaches to future assessment, such as forecasting and risk assessment, in that it explicitly considers a range of possible futures rather than focusing on the accurate prediction of a single outcome (van der Heijden 1996; Peterson et al. 2003).

The scenario planning process is typically conducted through a series of workshops involving scientists, managers and stakeholders such as non-governmental agencies, community groups, and members of the general public. Although not always attained, the shared understanding which these workshops may foster is often one of the most valuable outcomes of the process, and may itself be a reason for conducting a scenarios exercise (Evans et al. 2006; Zurek et al. 2008). Scenario planning can be applied at a wide range of scales. For instance, the Millennium Ecosystem Assessment (www.MAweb.org) developed scenarios at the global level (MA 2005b), as well as at regional and local levels (MA 2005a).
The scenario planning process may be characterized by a series of steps (Wollenberg et al. 2000; Peterson et al. 2003; Evans et al. 2006):

1. **Identify a focal issue.** To be effective, scenario planning should address a specific focal question (e.g., how will the ecosystem services provided by Lake Wingra change over the next three decades?). The focal issue is best identified with stakeholder input.

2. **Systems analysis.** An assessment of the people, institutions, ecosystems and their connections which define the system relevant to the focal issue. Based on this assessment, the major drivers (e.g. demographic change) and key uncertainties are identified (e.g., the degree to which green technologies become available).

3. **Brainstorm alternative trajectories.** The key uncertainties form the basis for brainstorming alternative ways in which the system could evolve. Based on this exercise, a set (usually 3-5) of alternative trajectories are identified that are particularly illuminating with respect to the focal issue.

4. **Build the scenario narratives.** Detailed stories linking the present to the future are drafted for each chosen trajectory. This is best done by a small group of individuals. Each story should track key indicators relevant to the initial focal question. The final stories should complement each other to form a coherent, thought-provoking set.

5. **Test the scenarios.** The draft storylines should be tested for consistency. The dynamics of the stories must be plausible. Consistency may be tested through interviews with stakeholders or by quantification.

6. **Use the scenarios.** Once the set of scenarios has been created they can be used to inform policy or further research.
Teaching through application: Developing scenarios for Lake Wingra

We explored the scenario planning process in a one credit graduate seminar with 13 students with backgrounds in the natural sciences and interdisciplinary social-ecological programs. We focused on the practical development of a set of scenarios for the future of Lake Wingra, a small urban lake in Madison, Wisconsin surrounded by substantial green space and used for swimming, fishing, and non-motorized boating. The scenarios were developed in collaboration with the Friends of Lake Wingra (FOLW), a community-based non-profit organization. The process was intended to help inform an FOLW-led initiative to develop a set of goals and associated management strategies for the Lake Wingra ecosystem.

The scenario planning group met once a week for two hours. Sessions were structured to alternate between reviews of published scenario exercises and methods, and the practical development of scenarios for Lake Wingra (see Appendix 1 for class schedule and readings). Early in the semester we had an opportunity to participate in a public meeting organized by FOLW at which goals for Lake Wingra were brainstormed. This gave us an understanding of the broader context to which the scenario planning process might contribute. Based on this understanding, the focal question (Step 1) for the scenario exercise was defined as: How will Lake Wingra and the ecosystem services it provides change over the next generation (to 2035)?

Next, through a process of brainstorming and discussion, we conducted a systems analysis exercise (Step 2) and developed a conceptual model of social and ecological elements that influence Lake Wingra (Figure 2). This analysis helped build appreciation and understanding
of the complex set of social and ecological drivers that affect Lake Wingra. Students found the process valuable as a practical example of systems thinking and in moving them outside their academic comfort zones.

Step 3 entailed the generation of alternative trajectories for the development of Madison that would have differing consequences for Lake Wingra. Alternative trajectories were brainstormed at a workshop with ten invited stakeholders from a variety of backgrounds. We explored trajectories in which it was very difficult or relatively easy for FOLW to achieve their goals. In addition, several “wildcard” storylines were brainstormed in which unexpected changes in drivers such as climate and urban population occurred.

The ideas generated at the stakeholder workshop were distilled to identify four key trajectories that, as a set, highlighted a handful of particularly illustrative and thought-provoking futures. A small group of students fleshed out narratives for these trajectories (Step 4), which were internally reviewed by the class. The revised scenarios were then discussed in one-on-one meetings with a variety of community stakeholders to obtain feedback on their plausibility and usefulness (Step 5). Based on this feedback the scenarios were again revised before being written up as a class report and formally presented to FOLW at a public meeting (Box 1).

**Reflections on the learning process**

Students provided detailed written reflections on the course eight months after completion of the exercise (see Appendix 2 for guiding questions). The course was widely regarded as
stimulating and fun (although a lot of work), and very different from other courses students had taken. All students felt they would recommend the class to others, and many expressed a desire to use scenario planning in their future careers. Based on a manuscript planning workshop held after students had reflected on the course, we agreed that the following four key learning points emerged from our experiences:

**Appreciating the prevalence of uncertainty**

Exploring possibilities for the future in a narrative framework forced students to expand their notion of uncertainty. One student commented that "*prior to this seminar, I mostly thought about uncertainty as error bars around a mean. Thinking about scenarios made me acknowledge the existence of qualitative unpredictability.*" Other students noted that the exercise highlighted that "*we are working in a world of many unknowns*" and that many uncertainties are driven by changing societal goals rather than scientific issues.

Several students felt the seminar changed their attitude towards uncertainty. One student observed "*People often treat uncertainty as a bad thing. Admitting uncertainty, and asking what policies make sense in the light of uncertainty is critical. Uncertainty deserves to be taken off the list of bad words.*" Another stated "*I felt encouraged... that although uncertainty is a great challenge, it can be worked with and incorporated into policies and management plans*." Although not all students felt as hopeful, most students recognized that scenario planning provided a different way of thinking about and dealing with uncertainty compared to traditional scientific approaches.
Working with non-academic stakeholders

"Eye-opening", "refreshing", and "inspiring" were how students described their interaction with non-academic stakeholders (people not working as scientists at academic institutions, e.g. government officials, agency scientists, local business representatives, community group representatives). Many students found the interaction with stakeholders the most valuable part of the class. In particular, it changed the way many students perceived those outside academia. For instance, one student observed "non-academic stakeholders demonstrated an impressive grasp of the issues...Their concerns did not always mirror those of scientists studying the lake. Many times their perspectives were holistic, encompassing ecosystem, watershed and community level thinking".

The importance of listening was a central lesson noted by students: "I learnt that while people feel strongly about a variety of things, most people are very willing to listen to academic viewpoints provided that academics are willing to listen to them". Students also felt they gained valuable verbal and writing skills, for example "I improved my writing by thinking outside of the research paper framework. I was challenged to write more clearly, concisely and with less jargon". One student realized in particular that "Engaging people in thinking about the future of our environment requires dealing in real stories, with color, and noise, and clutter".

Students felt motivated by contributing to a broader non-academic process: "The participation of non-academic stakeholders reminded us that the materials we were developing would be used by people outside the class... I think this helped to motivate class participants to produce work of high quality and complete work promptly". Other students noted that interaction with stakeholders "provided much-
needed 'real world' perspective for our academic work and gave us a better sense of the broader community in which we live as citizens.

Appreciating the need for multiple viewpoints

Scenario planning is inherently interdisciplinary. The systems focus of scenario planning substantially expanded students' perspectives. For example, one participant felt reminded that "ideas outside of ecology are important for thinking about the future of a lake and its watershed". Another student realized "I learned that for any environmental issue, each discipline feels like it has a solution to the problem...it was eye-opening to see how everyone's solutions were so specific to their disciplines".

The contribution made by non-academic stakeholders was especially illuminating. For example, one student realized that "science does play an important role in information gathering, but it is only one piece". Students also noted having learnt that "involving a diverse group of people brings necessary and different perspectives" and that "I have a better appreciation for all the different points of view that contribute information to any policy or management plan".

Better understanding the role of science in planning processes

Students found the scenario exercise valuable in exposing them to the complex linkages between science and policy. Prior to the scenarios course, many students felt cynical about the role of science in planning processes. For instance, one student believed that "policy was written and executed without being informed by science". At the other extreme were students who felt science played a driving role: "I had the somewhat idealistic view of science playing the strong role of
informant in planning processes and policy". By providing an example of how science can engage in a planning process, most students felt the course left them with a more nuanced, but positive view about the contribution which science can make in planning processes.

Many students specifically noted that the course changed the way they envisioned their role as scientists. For example, one student realized that with respect to his work "scenario planning could be used to empower people to create their own future". Another student, working on conservation issues in developing countries, felt that the storytelling approach in scenario planning "might prove valuable in transcending cultural barriers".

Conclusion and teaching challenges

Tomorrow’s environmental scientists will play a critical role in helping society navigate the environmental challenges of the 21st century (UN 2002; MA 2005c; IPCC 2007). Our experiences suggest that scenario planning can be an effective way of exposing graduate students to the post-normal scientific approaches needed to help address these challenges. As importantly, many students found exposure to post-normal approaches highly rewarding, encouraging and inspiring. If society is to address the environmental challenges we face, we need scientists who believe it is possible to deal with these challenges and who are inspired to engage society to find ways to do so.

We hope that our experiences may inspire educators at other institutions to offer similar courses. In this respect, several teaching challenges emerge. All students felt that the practical aspect of the course, and especially interacting with non-academic stakeholders, was
central to their learning. Finding a stakeholder group that has the interest and time to engage with a class of students may be a challenge. However, if a topic of mutual interest can be identified, our experience is that stakeholders find interaction as rewarding as the students do. Close interaction also improves the quality of the scenario products and increases their usefulness to the stakeholder group. A second challenge is that the value of a scenarios approach is often not evident to those unfamiliar with it. The notion of science as the provider of certain, factual information is deeply entrenched in modern culture, and it may be difficult to appreciate the value of a deliberate non-predictive exercise. Our advice is to approach the exercise as an experiment, drawing on inspiring cases of scenarios exercises (e.g., Kahane 1999; Wollenberg et al. 2000; Galer 2004) as examples of potential outcomes.

We also suggest several teaching improvements. Most students felt that the literature review components were useful but, given the limited time, would have preferred to spend more time on the practical exercise. In particular, students would have liked to interact more and with a wider variety of stakeholders. Many students felt the class would have been enriched by students with a broader range of disciplinary backgrounds. Both student and stakeholder diversity, however, entail trade-offs in efficiency. In most cases, aiming for mid-level diversity in both dimensions may be best. The degree of participation by stakeholders will largely be determined by pragmatic considerations, and successful exercises can likely be carried out with interaction at a range of levels. Lastly, many students felt that the class may have been more appropriate as a 3-credit course. An option for future instructors may be to offer a course at the 1-credit level focusing on a practical exercise, or as a 3-credit course including a broader review of the literature.
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References


Figure 1. Today's major environmental issues are characterized by the conditions of post-normal science: facts are uncertain, values in dispute, stakes high and decisions urgent. These conditions do not change the routine practice of traditional applied sciences and professional consultancy, but reframe the context in which they are carried out. Research problems become set and the solutions evaluated by a broader community of stakeholders. Figure adapted from Funtowicz and Ravetz (1993).
Figure 2. Conceptual model of the Lake Wingra system, spanning drivers at several spatial scales. Based on this analysis, we defined the spatial boundary for the scenario exercise as the City of Madison. This was the scale at which the most important interactions and feedbacks relevant to the Lake Wingra ecosystem were present.
Box 1: Lake Wingra scenarios in summary

We developed four scenarios describing the development of Madison city and the consequences for Lake Wingra and the ecosystem services it provides. The scenarios were built around four key uncertainties: 1) The level of environmental awareness and green technology at the national and global level, 2) The power and influence of grassroots environmental organizations within Madison, 3) Conflict between different user communities of Lake Wingra, and 4) Challenges posed by invasive species. None of the scenarios present an ideal or preferred outcome, highlighting the fact any development trajectory poses challenges and trade-offs. The full report is available at http://limnology.wisc.edu/courses/zoo955/spring2007/index.html.
Garden State: Propelled by concern for the global environment, enormous investments are made in green technology. These technologies, from solar roofs to biofuel feedstocks grown in residential rain gardens, permeate Madison. Local environmental groups are gradually assimilated by powerful global environmental organizations. The shift of influence to larger scales affects Madison's approaches to local environmental issues such as Lake Wingra. By 2035, the population of Madison has substantially increased and the lake system has become heavily engineered, leaving the long-term health of the lake in question.
**Big Green Brother:** Grassroots organizations transform government and divert funding to city-wide environmental projects. Strong steps are taken to address local environmental needs, including restoration of Lake Wingra. However, over time the new institutions become more narrowly focused and less responsive to evolving needs. The trend toward top-down management of local resources leaves a bad taste of big government in resident’s mouths. This rigidity meets a severe challenge when a deep, persistent drought strikes the Madison area. By 2035, tough choices have to be made between mitigating the effects of the drought and pursuing the goals of the Lake Wingra watershed.
C-Clear: Local organizations develop increasingly successful innovations for managing Lake Wingra. Use of the lake intensifies and the institutions representing the expanding user community become more diverse. However, conflict arises among the different interest groups, various coalitions form and political gridlock ensues. A spiraling cycle of emerging issues continually challenges those who wish to conserve Lake Wingra. By 2035, the ecological health of Lake Wingra has improved, but only a fraction of the stated goals have been met.
Exotic Exchange: Success in removing an exotic invader from Lake Wingra results in progress toward a healthier lake, but creates an ecological vacuum and exposes unexpected conflicts among user groups. In 2017 a new harmful invader fills the ecological vacuum and creates a new suite of problems for Lake Wingra. This catalyzes change and refocuses management efforts. By 2035, preventing future invasions has taken center stage and fish management has reoriented around the diverse interests of different user groups.
CHAPTER 6

CONCLUSION
Introduction

This dissertation employs a complex systems framework (Levin 1998; Miller & Page 2007; Norberg & Cumming 2008; Gros 2008) to address a diverse set of topical issues in ecosystem management, using a range of methods. Each of the core chapters (Chapters 2-5) are intended to stand on their own as peer-reviewed journal articles, and are all being prepared or are in the process of publication. Viewed together as a set of chapters in this dissertation, however, several cross-cutting themes emerge. This conclusion chapter briefly explores these themes.

Cross-cutting themes

Three key cross-cutting themes emerge from this dissertation: 1) Uncertainty in studying and managing complex ecosystems, 2) The need for integration across multiple domains to understand and manage complex ecosystems, and 3) The existence of policy windows and the need for proactive action in managing complex ecological systems. Each of these themes constitutes a major topic of study in their own right. The aim of this section is to briefly highlight key insights from and links between the core chapters with respect to these themes, rather than to address the themes comprehensively.

**Uncertainty in studying and managing complex ecosystems.** As outlined in Chapter 1, the nature of complex systems gives rise to uncertainty, particularly as a result of spatial heterogeneity, ongoing system evolution, and emergence. Chapters 2 and 3 provide specific insights with regard to the implications of uncertainty for studying and managing complex ecosystems. In Chapters 4 and 5, uncertainty is central to the context within which the
chapters are framed, but these chapters do not explicitly focus on the implications of uncertainty for studying and managing ecosystems. Rather, fostering transformation in ecosystem governance (Chapter 4) and teaching scenario planning (Chapter 5) can both be seen as responses or methods that may enable society to better deal with uncertainty associated with managing ecological systems.

Chapter 2 illustrates specific implications of the uncertainty associated with studying complex ecosystems in the context of environmental disputes. Using heuristic simulation examples, we show that different studies may often arrive at conflicting results due to environmental heterogeneity and the limitations of analytical tools. Because broader societal institutions for resolving environmental disputes are still largely rooted in a static "machine-like" view of ecosystems (Capra 1997; Cortner & Moote 1999), where it is expected that different studies should arrive at similar conclusions, the customary interpretation is that conflicting results increase uncertainty and that where studies disagree some studies must be flawed or fraudulent. We suggest instead that in the context of complex ecosystems conflicting results from different studies should often be expected. Rather than viewing conflicting findings as creating uncertainty, we suggest that valid, but conflicting findings are better viewed as enabling a broader assessment of the dynamics and uncertainties surrounding the issue in dispute. Generating such a broader view and enriched understanding requires methods that enable the integration of different studies and information sources, and is further discussed in the Integration cross-cutting theme.
In the context of environmental disputes, Chapter 2 further illuminates how traditional views about uncertainty can lead to protracted inaction. Most contemporary societal institutions for resolving environmental disputes are founded on the expectation that uncertainty can be reduced through further study, and that factual certainty needs to be established before action can be taken (Funtowicz and Ravetz 1993; Sarewitz 2004). In combination with the expectation that different studies should generally obtain similar results, these expectations often lead conflicting findings to result in stalled decision-making and action. Groups with vested interests in the status quo may capitalize on this dynamic by purposefully introducing conflicting information and thereby delaying action and change (Herrick and Jamieson 2001; Bocking 2004; Sarewitz 2004). As shown in Chapter 3, and further discussed under the Policy windows cross-cutting theme, this may have serious environmental consequences, especially where ecosystems are subject to ecological regime shifts. The occurrence of stalemate positions created by perceived uncertainty could potentially be reduced by a broader societal adoption of a complex systems understanding of ecosystem management, which would emphasize that uncertainty is inherent to complex systems, and in some cases irreducible. Furthermore it would highlight that there may be substantial costs associated with delaying policy action. Rather than expecting factual certainty to be established before taking action, decisions about whether or not to take action at a particular point in time should weigh the uncertainties associated with the need for action against the potential consequences of delayed action.

Chapter 3 investigates a different aspect of uncertainty in studying and managing ecosystems: the problem of detecting and averting unknown ecological thresholds. This chapter
highlights that current early warning indicators of ecological regime shifts cannot substantially reduce uncertainty about the location of an ecological threshold until an ecosystem has started undergoing a regime shift. If the factor driving the regime shift is amenable to rapid manipulation through management action, uncertainty about an approaching ecological threshold may be sufficiently reduced to provide warning that enables changes in management to avert a regime shift. However, if the factor driving the regime shift can only be manipulated gradually, current regime shift indicators provide insufficient warning to enable aversive management action to be taken. The implications of the fact that in ecosystems subject to regime shifts there is a point beyond which changes in management can no longer avert a regime shift is further discussed in the \textit{Policy windows} cross-cutting theme. In order to reduce uncertainty about the location of an ecological threshold with sufficient warning to adapt management to avert a regime shift, we conclude in Chapter 3 that critical levels of the regime shift indicators need to be defined, rather than simply focusing on detection of change in the indicators.

Both Chapters 2 and 3 suggest that a complex systems understanding of ecosystems requires the development of new societal institutions for addressing uncertainty in environmental disputes and informing policy action. Promising methods in this regard include joint fact finding (Andrews 2002; Schultz 2003; Karl \textit{et al.} 2007) and integrated assessments (Farrell & Jäger 2005; Mitchell \textit{et al.} 2006) such as the Millennium Ecosystem Assessment (MA 2003; MA 2005b). Such methods focus heavily on the assessment and integration of different sources of information, as further discussed under the \textit{Integration} cross-cutting theme. They generally involve extensive dialog between groups and individuals with different
perspectives, and are an important component introduced by transformations from sectoral, expert-centered approaches to more collaborative, integrated approaches to ecosystem management, as discussed in Chapter 4. Another increasingly popular method for acknowledging and assessing uncertainty is scenario planning (van der Heijden 1996; Peterson et al. 2003). Teaching scenario planning, which is the focus of Chapter 5, therefore represents another means for enabling society to better appreciate and deal with uncertainty in managing ecosystems. An underlying premise of scenario-planning approaches is that uncertainty needs to be explicitly acknowledged and incorporated into decision-making, and that action often needs to be taken in the face of uncertainty (Funtowicz and Ravetz 1993; Gibbons et al. 1994; van der Heijden 1996; Peterson et al. 2003).

The need for integration across multiple domains to understand and manage complex systems. Chapter 1 highlights that the nature of complex systems, especially the emergence of unique system properties at different scales, means that integration across multiple domains is required in order to understand and manage ecosystems. Particularly important in the context of ecosystem management is that, given the strong interactions that now exist between social and ecological systems, there is a need for study at the level of coupled social-ecological systems (SES) (Berkes & Folke 1998; MA 2005b). The chapters in this dissertation all adopt an SES perspective where both social and ecological aspects of the system, as well as their interactions, are seen as dynamic and evolving. Furthermore, the chapters highlight the benefits of integrating different sources of information in order to better understand and manage ecosystems, and introduce a variety of methods whereby such
integration can be achieved. Chapters 2 and 3 focus on the quantitative integration of information, while Chapters 4 and 5 focus on integration at a more qualitative level.

Chapter 2 shows how the quantitative integration of findings from different studies may enable the development of a richer and more nuanced understanding of ecosystem dynamics. In addition, the chapter emphasizes that the integration of conflicting information may represent a means for helping resolve disputes by incorporating and drawing on valid information put forward by all parties, rather than requiring choices to be made about which parties' information decisions should be based on. The ability to integrate multiple sources of information is an important issue in terms of the collaborative governance approaches discussed in Chapter 4. Similarly, Chapter 3 emphasizes the need to draw on multiple regime shift indicators in detecting ecological thresholds, rather than relying on a single indicator, as some indicators may perform better than others under certain conditions.

Chapter 3 further underlines the need to account for both slow and fast variables, in both social and ecological domains, in studying and managing ecosystems. In particular, while the immediate cause of a regime shift may be an environmental shock, the underlying cause is often gradual, unnoticed change in a slow variable. Such slow variables can be either social or ecological. A particular problem in studies that do not adopt an SES perspective, but take a predominantly ecological or social perspective, is that they may fail to pay attention to slow variables in the other domain. For example, an ecologically-focused study of eutrophication in a lake may fail to account for gradual social or economic changes that influence land use in the lake's watershed, while a socially-focused study of the same issue may ignore the slow
build-up of phosphorous in soils. These slow changes may lead to longer-term effects unanticipated by the more narrowly focused studies. The results in Chapter 3 imply that the ability to anticipate and avert regime shifts may be critically dependent on monitoring and assessing the consequences of changes in both social and ecological variables.

Together, Chapters 2 and 3 highlight the need for further development of quantitative and qualitative methods for integrating multiple information sources and perspectives. Chapter 2 introduces one relatively new quantitative integration method: hierarchical Bayesian techniques. Chapters 4 and 5 involve more qualitative approaches to integrating information. Chapter 4 focuses on factors that may facilitate transformation from more sectoral, expert-centered approaches to more integrated, collaborative approaches to ecosystem governance. Such collaborative approaches rely heavily on techniques that facilitate dialog and enable the acknowledgement and integration of a variety of perspectives in decision-making (Gray 1989; Isaacs 1999; Kahane 2004; Senge et al. 2004). Chapter 4 suggests that fostering environmental awareness, developing leadership capacity, facilitating dialog between key parties, and providing institutional support are important in enabling more collaborative forms of governance to emerge.

Chapter 5 involves the use of scenario-planning, which takes a systems approach to understanding future trajectories of change in coupled SES. Scenario-planning approaches are highly integrative, and draw on diverse sources of information, values and perspectives in developing plausible alternative development trajectories. Although not undertaken in the context of Chapter 5, scenario-planning also represents a powerful tool for integrating
qualitative and quantitative information. For instance, the Millennium Ecosystem Assessment first developed integrative, qualitative storylines of alternative trajectories of future changes in the world's ecosystem services. These storylines were then quantified using large integrated assessment models such as the Integrated Model to Assess the Global Environment (IMAGE) (Bouwman et al. 2006), and the results were used to revise the qualitative storylines (MA 2005a). As such, scenario-planning can be a powerful method for integrating knowledge from different arenas, such as scientific knowledge and more tacit forms of knowledge held by experts, indigenous peoples and the general public.

The existence of policy windows and the need for proactive action in managing complex ecological systems. As highlighted in Chapter 1, complex systems can often exist in multiple regimes, where shifts between regimes usually involve large, abrupt changes, and may be difficult or impossible to reverse. Insights from Chapters 2-4 have particular relevance for society's ability to take action to avert undesirable regime changes.

Chapter 3 clearly illustrates that there exist discrete windows in which policy action can bring about the biophysical changes necessary to avert ecological regime shifts. Chapter 3 illustrates that the policy windows for variables that can be rapidly manipulated through policy action are considerably greater than for variables that can only be gradually manipulated. In particular, if variables are amenable to rapid manipulation, action to avert a regime shift may be delayed until a regime shift is well underway, whereas if variables can only be gradually manipulated action is required substantially before a regime shift if it is to be averted. Chapter 3 has important implications for the monitoring of ecological variables,
and the expectation that policy changes should be informed by observed changes in ecological conditions. In particular, this chapter highlights that there may be little observable change in ecological conditions or regime shift indicators before the point at which the biophysical window for policy action ends. As outlined in the preceding discussion, this underlines the need to be cognizant of gradual changes in driving variables (e.g. angling or shoreline development) that may affect longer-term ecological conditions. In addition, it emphasizes the need to integrate different sources of information, in this case monitoring information with ecological models, in order to appropriately inform management action.

Chapter 4 on the other hand emphasizes the existence of sociopolitical windows in which it is socially and politically practical and feasible to introduce and implement policy changes. This chapter highlights that such sociopolitical policy windows are often created by environmental crises. In this respect it suggests that biophysical and sociopolitical windows for policy action may often be mismatched: the biophysical window for effective policy action may close before the socio-political window that makes policy action feasible is opened. As noted in the following section, research into better matching biophysical and sociopolitical windows for policy action is therefore an important topic for further research.

The type of ecosystem governance transformations that are the subject of Chapter 4 raise additional concerns about the ability of society to respond proactively to environmental degradation, and avoid undesirable regime shifts. The integrated, collaborative governance approaches discussed in Chapter 4 can result in more protracted decision-making processes than more authoritarian, expert-centered approaches. However, this may only be true where
there is not strong opposition to authoritarian approaches; where strong opposition exists the ensuing disputes may lead to substantial delays in decision-making and action compared to successful collaborative approaches. In general, it may be that authoritarian approaches are more suitable and acceptable in the face of crises (e.g., restrictions imposed in response to the Severe Acute Respiratory Syndrome (SARS) outbreak in 2002/2003). Under non-crisis conditions, modern Western societal values around democracy and self-determination may mean that collaborative approaches are more effective at creating socio-political policy windows that enable durable policy action.

Chapter 2 highlights a further dynamic that may restrict the creation of socio-political windows for policy action. As discussed under the Uncertainty cross-cutting theme, the expectation that uncertainty has to be resolved before policy decisions can be made can create an incentive for groups with vested interests in the status quo to introduce conflicting information that ostensibly increases uncertainty. As highlighted in the earlier discussion, the adoption of a complex systems understanding emphasizes the need to weigh up the uncertainty associated with the need for action against the potential costs of delayed action in environmental decision-making. This implies that rather than aiming to resolve uncertainty before taking decisions, a more effective approach to managing complex ecosystems is to continually assess uncertainty and make decisions about whether to implement policy changes or not in the context of that uncertainty. Scenario-planning, discussed in Chapter 5, is one method that can facilitate assessment of uncertainty and decision-making in the face of uncertainty. For instance, the different scenarios developed during a scenario-planning process can be used to assess the robustness of alternative policy
options under different potential future conditions (van der Heijden 1996; Peterson et al. 2003). This can help inform policy decisions that reduce potential regrets.

Future research areas

The cross-cutting themes discussed in this chapter highlight several potentially fruitful topics for future research:

♦ **Better assessing and dealing with uncertainty in ecosystem management.** The insights from this chapter underscore the need for improved methods and procedures for assessing uncertainty in SES dynamics. It further emphasizes the need for developing societal institutions that are better able to deal with uncertainty in environmental decision-making, and particularly to weigh up uncertainty around the need for action against the potential costs of delayed action.

♦ **Methods for better integrating different sources of information.** This chapter highlights the need for improved quantitative and qualitative approaches that enable different sources of information to be integrated and synthesized. Using more integrated sources of information in environmental decision-making in many cases also requires the development of new and modified decision-making procedures in society.

♦ **Better assessing and matching biophysical and sociopolitical windows for policy action.** Insights from this chapter emphasize the need for methods that enable better assessment of biophysical windows for policy action, and procedures that facilitate the
matching of sociopolitical and biophysical windows for policy action. Better matching sociopolitical and biophysical windows for policy change will in many cases rely on development of institutions that enable society to respond more promptly to warnings of undesirable environmental changes.

References


