especially in cases of surprise, that is, when social–ecological systems behave in unforeseen ways. Chapter 3 reviews and assesses the impact of resilience thinking on the way social–ecological systems have been conceptualized. Chapter 4 provides a boldly speculative perspective that cuts across disciplinary boundaries to explore the implications of redundancy in its various forms.

2

Adaptive dancing: interactions between social resilience and ecological crises

LANCE H. GUNDERSON

2.1 Introduction

Systems of people and nature co-evolve in an adaptive dance (Walters, 1986). Resource systems change as people seek ecosystem services, such as the harvest of stocks, manipulation of key structuring processes, removal of geophysical assets or abation of pollutant concentrations. Meanwhile, as humans are becoming more dependent on these ecosystem services, the ecosystems become more vulnerable to unexpected events. This process that signals a loss of ecological resilience has been described as a pathology of resource development (Holling, 1995).

Complex resource systems are not easily tractable or understood, much less predictable. Nonlinear interactions among multiple variables, scale invariant processes, emergent properties from self-organization and other factors all contribute to unpredictability. Yet, even with these inherent difficulties, we continue attempts at making sense for management and other purposes. Due to a growing empirical base of observation, emergent patterns of these systems, including periods of stability and instability, as well as unexpected behavior due to internal and external changes have been revealed (Gunderson, Holling, and Light, 1995; Berkes and Folke, 1998; Johnson *et al.*, 1999).

This paper builds on earlier work (Holling, 1978; Walters, 1986, 1997; Gunderson *et al.*, 1995; Gunderson, 1999a) to explore these unexpected behaviors in managed ecological systems – perceived as surprises and crises. To begin with, the conceptual basis for understanding these nonlinearities, ecological properties of resilience and adaptive capacity, and analogous properties in institutions are presented. The next section describes a set of different types of surprises, followed by a discussion of how people respond to those different types of surprises. The chapter ends with some tentative propositions on how one might move beyond sense-making and begin to manage for resilience.

2.2 Resilience in ecological and social systems

Complex resource systems link ecological components and social components (including economic systems, institutions, and organizations). Institutions are described as the set of norms, rules that people use to organize activities (Ostrom, 1990). Social systems are comprised of three types of structures (signification, domination, and legitimation) that enable power and resources distributions, patterns of authority in addition to norms, rules, routines and procedures (Giddens, 1987). At the heart of these components and their interaction are the properties of resilience and renewal. Resilience provides these complex systems with the ability to persist in the face of shocks and disturbances. Maintaining a capacity for renewal in a dynamic environment provides an ecological buffer that protects the system from the failure of management actions that are taken based upon incomplete understanding, and therefore allows managers to affordably learn and change.

Resilience of a system has been defined in two very different ways in the ecological literature, each reflecting different aspects of stability (Fig. 2.1). Holling (1973) first emphasized these different aspects of stability to draw attention to the tensions between efficiency and persistence, between constancy and change, and between predictability and unpredictability. The more common definition considers that ecological systems exist close to a stable steady-state. In this context, resilience is described as a return time to a steady-state following a perturbation (Pimm, 1984; O'Neill *et al.*, 1986). This definition has been described as engineering resilience (Holling, 1996) and carries an assumption of a single, global equilibrium (Fig. 2.1A).

The second definition emphasizes conditions far from any stable steady-state, where instabilities can flip a system into another regime of behavior, i.e., to another stability domain (Holling, 1973). In this case resilience is defined as the magnitude of disturbance that can be absorbed before the system redefines its structure by changing the variables and processes that control behavior. This is termed ecological resilience (Walker *et al.*, 1969), as depicted in Figure 2.1B. Those who emphasize the stability domain definition of resilience (i.e., ecological resilience), on the other hand, come from traditions of applied mathematics and applied resource ecology at the scale of ecosystems, e.g., of the dynamics and management of freshwater systems (Fiering, 1982), of forests (Clark, Jones, and Holling, 1979), of fisheries (Walters, 1986), of semi-arid grasslands (Walker *et al.*, 1969), and of interacting populations in nature (Dublin, Sinclair, and McGlade, 1990; Sinclair, Olsen, and Redhead, 1990).

Recent advancements suggest that a third category is needed to describe ecological change. In the above-mentioned definitions of resilience, both are based

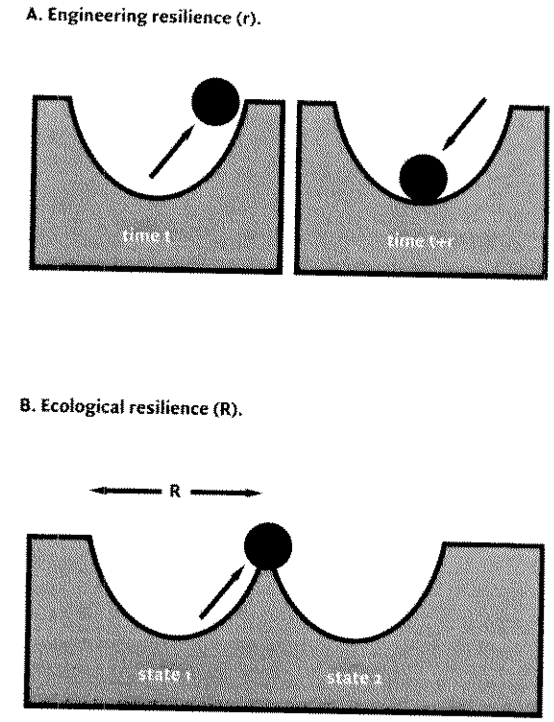

Figure 2.1 Alternative definitions of resilience as represented by a ball and cup model. Cups represent the stability domains of the system, the ball represents the system state, and single arrows represent disturbances to the system. (A) Engineering resilience can be depicted by a global equilibrium (ball resting at the bottom of a cup). When the system is disturbed (ball moves up the side of the cup), resilience is defined as the amount of time (r) for the system to return to the equilibrium state. (B) Ecological resilience is defined as the amount of disturbance that the system can absorb without changing state (stable state 1 or 2), and is measured as the width of the stability domain (R).

on the notion of a system with a stationary stability domain. The structures and processes that produce stability are assumed not to change over time or space, and hence are tractable. In the return-time concept, a single stability domain is implicit, whereas in the ecological resilience concept, multiple steady-states are possible. Yet the kinds of ecological processes that create these stability basins are slowly changing variables: mud in lakes (Carpenter, Ludwig, and Brock, 1999b); species composition in semi-arid rangelands (Walker *et al.*, 1969); soil nutrient concentration in wetlands (Davis, 1994); or spatial connectivity of old trees in spruce budworm forests (Ludwig, Jones, and Holling, 1978). Hence, another term is needed to describe the capacity of a system to adapt to these slower dynamics – described as adaptive capacity (Peterson, Allen, and Holling, 1998; Gunderson, 2000).

When ecosystems are observed to shift in behavior, structure or functions, it is usually signaled as a resource crisis. The literature is replete with examples: sudden blooms of toxic algae in freshwater lakes (Carpenter, Brock, and Hanson, 1999a), emergence of shrubs in semi-arid grasslands (Walker *et al.*, 1969), shifts in species dominance in freshwater wetlands (Davis, 1994). A weak typology of different types of surprising ecosystem behaviors is described in the next section.

2.2.1 Ecological surprise

In these co-evolving systems of humans and natures, surprises are the rule, not the exception. An ecological surprise is defined as a qualitative disagreement between ecosystem behavior and *a priori* expectations – an environmental cognitive dissonance. Brooks (1986) provides a useful typology of surprises in describing the interaction between technology and society, and defines three types; (1) unexpected discrete events, (2) discontinuities in long-term trends, and (3) emergence of new information. These categories can be broadened and placed in the context of the previously mentioned theories of change in ecological systems, by redescribing Brooks' (1986) types as local surprise, cross-scale surprise, and true novelty. Examples and elaboration are described in the next paragraphs.

Local surprises can often be addressed by recognizing broader-scale processes. Unexpected discrete events can be part of broader-scale fluctuations or variation of which there is little or no local knowledge. An ecological example of this is the cycle of flood and drought over the southeastern USA, which is part of global atmospheric and oceanic coupling known as El Niño/Southern Oscillation (ENSO). In these cases, the ignorance of broader and longer-term processes and human limits on perception both contribute to the local surprise.

The next class of surprise deals with abrupt, nonlinear or discontinuous behavior of a system that, after analysis, can be attributed to an interaction between key variables that operate at distinctly different scale ranges. That is, the surprise is due to a faster variable interacting with slower variables. In ecological systems, examples include spatially contagious processes, such as forest fires, which only occur when there is an interaction amongst a trigger such as a spark, dry fuel, and sufficient fuel for the fire to carry. In this example, ignition frequency (such as lightning strike rate) is a 'fast' variable and the spatial distribution and amount of fuel characterize more slowly changing variables

(Holling, 1986). In these dynamics lay the interactions for qualitative shifts in stability domains of resource systems, such as Walker *et al.*'s (1969) analysis of subtropical savanna grazing systems, or in the dynamics between stable states of lake systems (Scheffer 1998; Carpenter *et al.*, 1999b).

The final type of surprise is genuine novelty – that is, something truly unique and new or not previously experienced by humans (or at least outside the breadth of captured experience for a culture in a new situation). These types of surprise can generate change, the consequences of which are inherently unpredictable. Most examples of new technologies (such as the personal computer in the late 1970s) fall into this category (Tenner, 1996). In resource systems, invasions by exotic species are this type of surprise. The invasion of alien trees such as *Myrica faya* in Hawaii (D'Antonio and Vitousek, 1992) or *Melaleuca quinquenervia* in Florida (Myers, 1983) alters key ecosystem processes such as nutrient cycling, water relations, and fire patterns. Perhaps the greatest surprises ahead of us are the unforeseen planetary impacts of humans, such as facilitated species movement, nutrient cycling changes, changes in land-use, and the creation of new substances.

2.2.2 Ecological crises

Ecological crises are a special type of surprise. That is, a surprise becomes a crisis when it reveals an unambiguous failure of policy. As elsewhere (Gunderson *et al.*, 1995), the term *policy* is used in this context to describe the rules, norms, behavior, and infrastructure of management action (Ostrom, 1990).

Not all ecological surprises lead to crises of policy. In situations where flexibility in policy exists, variations in external events can be easily managed without a policy change. Examples of this would include adaptive responses of fire management agencies to the outbreak of fires that were associated with ENSO fluctuations. In 1998, fire management officials in Florida were able to manage severe drought and fire conditions by calling on a large pool of firefighting resources to deal with unexpected fire outbreaks. In other surprises, such as the massive die-off of seagrass in Florida Bay in the late 1980s (Robblee *et al.*, 1991), little or no change in policy can occur, even though they were viewed as ecological crises. The point here is that some surprises can be managed, without leading to a policy change, whereas others result in shifts in policy. There is further discussion of this dichotomy later in the chapter.

On the other hand, not all crises stem from ecological surprises. Shifts in ecological policy can be due to broader social reforms or changes in the way people view or understand resource issues. Light, Gunderson, and Holling (1995) indicated a major reformation of policy in Everglades water management in the early

1980s due to a shift in public perception of water quality concerns, when little or no ecological change had been documented. Costanza and Greer (1995) suggest that policy reformations of water quality monitoring and action in Chesapeake Bay were rooted in a changing sense of stewardship and responsibility by key people (activists, legislators, and scientists) who lived on the bay.

Rather than try to understand the role of crises and surprises in a broad set of conditions, this chapter focuses on how people respond to those ecological crises that signal the erosion of resilience and a shift of stability domains. In a review of large-scale ecosystems (Gunderson and Pritchard, 2002), we found that at least three different pathways can lead to the loss of resilience and inevitable surprise. The first pathway involves the addition of key substances into the ecosystem. Examples include the addition of low levels of phosphorus into the Everglades wetlands (Davis, 1994) or lakes (Carpenter and Cottingham, 1997). The second pathway involves removal of key resources or sources of resilience. Examples of these include the removal of soil in tropical forests (Lugo *et al.*, 2002), or removal of drought-tolerant plant species via overgrazing (Walker, 1995). The third pathway involves the manipulation of keystone ecological processes by human intervention, be they self-organized spatial patterns of forests with fire (Peterson *et al.*, 1998) or budworm (Ludwig *et al.*, 1978) or of key trophic relationships in coral reefs (McClanahan, Done, and Polunin, 2002). In each of these groupings, the differences in mechanisms that led to a loss of resilience created different circumstances for understanding and action during and after the ecological crises. How humans respond to these crises and make sense of the unexpected shift in stable states of nature is the topic of the next section.

2.3 Responding to crises

As Thompson (1983) states, we have no escape from having to 'manage the unmanageable.' Given that humans will continue to cope with systems that are partly knowable and partly unknowable, the ways in which people begin to make sense and develop dynamic responses are linked to the types of surprises and crises. The relationship between different types of uncertainty is key: how people choose to deal with uncertainty appears to either increase or decrease the resilience of an ecosystem. It is the ecological resilience that allows managers a margin of failure. There is growing evidence that acknowledgement and confrontation of uncertainty add resilience to managed systems (Gunderson, 1999a). But the subtleties and nuances of how that uncertainty is managed are complicated and not very well understood.

People involved in the practice of resource management are all linked by the need for understanding. Yet in these complex resource issues, uncertainty is

pervasive. Partitioning that uncertainty is an initial step for an approach that involves confronting and the hope of winnowing. Lee (1993) recognized that resource uncertainties can be separated into categories of social uncertainty (agreement on social objectives, norms) and technical uncertainty (understanding and explaining the mechanisms associated with a resource issue). Resolving the technical uncertainties of resource issues has generally been the domain of a technical and expert community, as described in the following section. Researchers such as Williams and Matheny (1995) and Pritchard and Sanderson (2002) suggest that the social uncertainty category of Lee (1993) is generally addressed by two groups: a political community and a stakeholder community. The second section that follows describes how social systems (including stakeholders and political components) manage uncertainty.

A key unresolved issue, however, is how these three communities interact (or not) around notions of ecosystem management and resilience (Pritchard and Sanderson, 1999, 2002). Often, what is most critical is the management of uncertainty among the three communities (technical, stakeholder, and political). A brief discussion of the interaction among technical, stakeholder, and political communities is presented in the third section that follows on integrating activities.

2.3.1 Technical and expert community

In these complicated systems, a great deal of uncertainty has been assigned to the technical and expert community for resolution. The experience and practice during this century have been to turn to scientists, the heart and soul of technology and technologic solutions, as the fountains of understanding. But there has been a growing sense that traditional scientific approaches are not working, and indeed make the problem worse (Ludwig, Hilborn, and Walters, 1993). Two reasons why rigid scientific and technological approaches fail are because they tend to focus on the wrong types of uncertainty and on narrow types of scientific practice. Many formal techniques of assessment and policy analysis presume a system near equilibrium, with a constancy of relationships, and that uncertainties arise not from errors in tools or models, but from lack of appropriate information on which to base the models. Walters (1997) outlines the challenges faced by the scientific and technical community in attempting to assess and resolve uncertainties of resource issues, including lack of data across appropriate scales and difficult cross-scale modeling problems, among others.

A conflict also arises between two views of science that contributes to perpetuate uncertainty rather than winnow it. One mode of science focuses on parts of the system and deals with experiments that narrow uncertainty to the point of acceptance by peers; it is conservative and unambiguous by being incomplete and fragmentary. The other view is integrative and holistic, searching for simple structures and relationships that explain much of nature's complexity. The latter view provides the underpinnings for an approach to dealing with resource issues called adaptive environmental assessment and management (Holling, 1978; Walters, 1986), which assumes that surprises are inevitable, that knowledge will always be incomplete, and that human interaction with ecosystems will always be evolving.

Another common pathology in environmental assessments involves the level of complexity in analysis and explanation. Many of these assessments engage scientists and technical experts to compile existing information in two forms: one is a set of facts or observations about the status of the resource and the second is in generating a set of plausible explanations about what has generated the ecological surprise. Accounts of recent examples indicate that the assessments generally reveal a paucity of reliable data. Sapp (1999) describes the generation of large research and monitoring activities around the world following the coral reef crisis in the 1960s associated with population outbreaks of the crown-of-thorns starfish. Similar results occurred following the seagrass die-off in Florida Bay in the late 1980s (Robblee *et al.*, 1991). Whereas a common response is to increase monitoring activities in order to learn more about the ecosystem, a more interesting set of activities deals with how the technical community attempts to sort through competing explanations or hypotheses that explain the crisis.

Alternative sets of hypotheses can always be generated to explain an ecological crisis. Using the examples just mentioned, at least four hypotheses were originally proposed for the crown-of-thorns outbreak: increased dredging, warmer sea temperatures, typhoons, and nuclear fallout (Sapp, 1999). In the Florida Bay example, at least seven hypotheses were proposed (Robblee *et al.*, 1991; Gunderson and Walters, 1999), including hypersalinity, excess nutrients, diseases, loss of grazers, lack of hurricanes. In each of these cases, the construct involved a single variable relationship. That is, one factor was supposed to have exceeded a level to which the ecological system was pre-adapted. Psychologists argue that in part it is due to cognitive limits (Dörner, 1996). In both examples (and in others), what generated a more successful explanation was the development of frameworks (models) that allowed two things: a rigorous comparison between data and hypotheses, and models that contained a minimum complexity. This debate still goes on, but there is evidence from these examples and others that a small set (more than three and less than five) of structuring variables can be used to explain much of these dynamic patterns

and interactions (Holling, Gunderson, and Peterson, 2002). This was certainly the case in the crown-of-thorns outbreaks (Bradbury, 1990) and the Florida Bay seagrass die-off (Gunderson and Walters, 1999).

In successful assessments, the models and data are used to exclude or invalidate hypotheses, leaving the remaining ones to be put at risk through management actions. Yet that too can be problematic, especially when different actions are required to sort among hypotheses (Walters, Gunderson, and Holling, 1992; Gunderson, 1999b). For example, in Florida Bay, the alternative explanations each suggested a different management strategy and action to attempt invalidation, along with varying social and political trade-offs. If the salinity hypotheses were true, then actions aimed at controlling nutrient inputs would be a costly misappropriation of resources.

Assessments and understandings can also be linked to the nature of the surprise and the mechanisms associated with the loss of ecological resilience. Understanding an ecological crisis that arose from a local surprise is perhaps the most tractable. Longer-term data sets and broader spatial perspectives both allow for linking the crisis to variables that are distant in space and time. Examples of these include the increased understanding of the role of ENSO (Glantz, Katz, and Nicholls, 1991) in driving local crises. Another example is that the recruitment patterns of key fish stocks in the Baltic are linked to weather systems in the north Atlantic (Jansson and Velner, 1995). Even the cross-scale surprises are becoming more tractable not just through increased data collection, but also through more sophisticated modeling. Many of the crises and surprises noted elsewhere (Gunderson *et al.*, 1995, Gunderson and Walters, 1999; Sapp, 1999) were resolved to the point at which actions could be initiated by this iteration between models and data. It is the true novelties that will continue to be the most intractable of the surprises.

Many authors argue that the failures of agency-based resource management and the abilities of those agencies to respond to ecological crises are in part due to technical limitations (Walters, 1997). Yet that community is often embedded in and operates within multiple sets of other human-dominated systems. A brief discussion of those communities and how they deal with different types of uncertainty is the subject of the next paragraphs.

2.3.2 Social systems

A unique property of social systems in response to uncertainty is the generation of novelty. Novelty is key to dealing with surprises or crises. Humans are unique in that they create novelty that transforms the future, and often it is the ecosystem crisis that spawns brief periods of creativity.

Often, new types and arrangements of management institutions are created after resource crises. These institutions can be formal, government-based agencies, such as the State of Florida Water Management Districts which were created following the drought of 1971 in the Everglades (Light *et al.*, 1995). Other formal agencies can span gaps in existing governance, such as the interstate compact that created the Northwest Power Planning Council in the Columbia River Basin (Lee, 1993). Often, scientific-based epistemic communities arise, such as the Baltic Management Commission in the Baltic Sea (Jansson and Velner, 1995), or a similar group in the Mediterranean (Haas, 1990).

The epistemic communities often include a broad spectrum of views, and provide a forum well suited to address surprises. Thus, one interpretation of these arrangements is that these institutions are set up to resolve different types of uncertainties. They provide a venue in which some technical and social uncertainties can be resolved (Lee, 1993; Pritchard and Sanderson, 1999; Pritchard, Folke, and Gunderson, 2000).

Yet there are many situations in which the institutions constantly struggle with resolving those uncertainties. That conservatism or institutional inertia can be described as an inability to re-invent themselves and adapt to changing conditions. Nor do agencies appear capable of generating either novel solutions to or policies for chronic resource issues. Indeed, one of the few mechanisms for agency change is the advent of an ecological crisis, as has been argued elsewhere (Gunderson *et al.*, 1995). But there are many other reasons why social systems are so conservative and inflexible regarding policy changes.

One reason why management institutions have such high moments of inertia is that they utilize (directly or indirectly) ambiguities and uncertainties of resource issues to maintain a *status quo*. With a pragmatic focus on policy implementation, most agencies seem to have a two-fold strategy that is aimed at reinforcing the *status quo*: prove that extant policies are wrong, and do not act until one is confident of what to do next. Many agencies focus on implementation, without realizing that narrow implementation schemes often subvert policy intent, or realizing that implementation is an organic process that changes over time and reveals the failure of policy, not its success (Gunderson *et al.*, 1995).

Another conserving strategy is seen by vested interests that have political and social sway over agencies. Whereas science uses uncertainty to drive the engine of inquiry, vested interest groups use and foster uncertainty to maintain a *status quo* policy. There are many examples – take the actions of sugar farmers in the Everglades following claims that nutrient run-off was changing the structure and function of pristine areas in the Everglades. Prominent scientists were hired to generate alternative hypotheses (other than those that involved

phosphorus), which for a while stalemated any movement towards resolving the crisis. Similar results of dis-information campaigns are chronicled for health, climate change, and biodiversity issues (Ehrlich and Ehrlich, 1996). Vested interests are not the only groups that generate or defend pet hypotheses. Agency scientists often generate policy recommendations that are politically correct in the sense of gaining what they view as a favorable policy. Take the Florida Bay example mentioned earlier. Most agency scientists gravitated to the freshwater run-off–salinity hypotheses as the reason for the large-scale seagrass crisis. It held political sway to the point where extra water was indeed delivered to Everglades National Park and Bay, with the counterproductive result of regeneration being delayed rather than accelerated (Walters, 1997). These examples further highlight the point that science is a highly social process, with lots of tacit and implicit factors influencing and shaping an 'objective' process.

2.3.3 *Integrating activities*

Currently, there is a lot of activity aimed at more integration of agencies, stakeholders, and citizens with regard to resource issues (Westley, 1995; Pritchard and Sanderson, 2002). Community-based resource management, citizen science, amongst others, are common buzz words and approaches. Examples include federal task forces or sustainability commissions that attempt to invite and engage all interested parties to help resolve the chronic resource issues. There are some examples, e.g., Florida Everglades (Gunderson, 1999b) and the Mediterranean Sea (Haas, 1990), of these meshing groups helping to link formerly disparate communities of agencies, stakeholders, and citizens around the issue of ecosystem restoration and resolution of chronic issues. These groups often fill 'structural holes' in problem domains, i.e., where explicit or implicit partitioning of responsibilities still leaves some uncertainties not addressed. These meshing functions appear to be a robust solution to filling obvious and critical gaps to resolve these complex issues.

Yet, in terms of resolving chronic issues or breaking through the types of inertia previously mentioned, most of the seemingly successful integrating activities appear to arise from temporary groups. The history of the Everglades provides an example. Following the drought of 1971, the Governor of the state of Florida called a symposium of the experts on how to deal with water supply issues. The convention created the design of water management districts in the state (Light *et al.*, 1995). The convention was then dissolved, passing the charge to resolve water supply and flood issues to a formal institution. The Everglades also provide examples in which formalizing these types of organizations was unsuccessful. Numerous technical committees have been formed to deal with resolving uncertainties that required integration across scientific disciplines or even between science and political arenas. Yet history indicates that these formal temporary groups tend to resolve little, if anything, of an issue that they were established to address. So this inconsistency raises a paradox: how to sustain a necessary role for integration across agency boundaries and across scientific disciplines, when most of the seemingly successful groups are temporary. This paradox is addressed in the next section, with some suggestions on how to create more sustaining types of institutions and inter-organizational activities.

2.4 Surfing ecological crises

2.4.1 Learning-based institutions

Perhaps it is time to rethink the paradigms or foundations of resource management institutions, and to place more emphasis on the development of sustaining foundations for dealing with complex resource issues. Learning is a long-term proposition, which requires a ballast against short-term politics and objectives. Another shift will probably require a change in the focus of actions away from management by objectives and determination of optimum policies, towards new ways to define, understand, and manage these systems in an ever-changing world. That focus should not be solely on the variables of the moment (water levels, population numbers) and their correlative rates, but rather on more enduring system properties such as resilience, adaptive capacity, and renewal capability. This framework involves both the human components of the system (operations, rules, policies, and laws) and the biophysical components of the landscape and its ecosystems. The shift of focus to learning basis is likely to require flexible linkages with a broader set of actors or network. Another way of saying this bluntly is: until management institutions are able and willing to embrace uncertainty and systematically learn from their actions, adaptive management will not continue in its original context, but rather will be redefined in a weak context of 'flexibility in decision-making' (Gunderson, 1999b).

In order to meet these challenges for learning-based institutions, there is a need to develop new theory and expand old theories addressing issues of scale. In times of uncertainty, there is nothing more practical than theory (Holling, 1995). Walters (1997) cites the cross-scale problem as a severe obstacle in most assessment/modeling activities. Development of new theories is needed to help address ecosystem and natural resource dynamics across space and time scales. Over the last 40 years, time and space have been separated for analytical purposes. Most field-scale ecologic investigations either freeze space and experiment over time or freeze time and look at spatial patterns (witness the

explosion and ubiquity of Geographic Information Systems (GIS) technology in resource management agencies). Perhaps there are practical reasons for this pattern, but also it can be explained in part because of underlying theoretical frameworks. There is a growing sense that this separable-dimension framework will result in different outputs from assessments, suggesting the need for integration.

There is a growing trend towards addressing the issues and theories of cross-scale interactions in ecology (Levin, 1992). These frameworks include the types of resilience arguments mentioned earlier, with multiple stability domains in various systems, and the controls among such domains (Gunderson *et al.*, 1997), and patterns of cross-scale discontinuities – where textural discontinuities in cross-scale structures create templates or signatures of similar lumpiness in animal community structure (Holling, 1992).

2.4.2 Can we manage for resilience?

In addition to developing better theories and trained professionals, perhaps we should attempt to develop new paradigms or schema that underlie resource management approaches. One place to start is with the notions of resilience and adaptive capacity. These theoretical concepts identify at least two strategies: those that people employ in order to manage for resilience in a resource system and properties that contribute to flexibility in human organizations. In order to add resilience to managed systems, a number of strategies are employed: increase the buffering capacity of the system, manage for processes at multiple scales, and nurture sources of innovation and renewal, as elaborated in the following paragraphs.

Most activities for buffering tend to address the engineering type of resilience, that is, mitigating the effects of unwanted variation in the system in order to facilitate a return time to a desired equilibrium. In many agricultural systems, resistance to change is dealt with by a combination of barriers to outside forces (tariffs, fences, etc.) and internal adjustments such as cost control mechanisms (Conway, 1993). Water resource systems can be designed for resilience by increasing the buffering capacity or robustness through redundancy of structures (and flexibility of operations) rather than fewer, larger structures and rigid operational schemes (Fiering, 1982). Berkes and Folke (1998) suggest that traditional approaches (which they define as traditional ecological knowledge) buffer managed systems by not allowing unpredictable or large perturbations to threaten ecosystem structure and function by allowing smaller-scale perturbations to enter the system. One such example is the Cree fishers' use of a mixed-size mesh net to harvest multiple age classes, thereby preserving an age-class structure that mimics a natural population (Berkes, 1995). This stable age structure helps buffer widely varying reproductive success.

Resource systems that have been sustained over long time periods increase resilience by managing processes at multiple scales. Returning to the example of the Cree in northern Canada, Berkes (1995) argues that multiple spatial domains are part of their fishing practices and multiple temporal domains in their hunting. While fishing, the Cree monitor catch per unit effort. When they notice the rate dropping, they immediately move to alternative fishing sites; over longer time frames, they rotate fishing effort to more remote sites.

Another example is in the Everglades water management system, where, in the mid 1970s, water deliveries to Everglades Park were based upon a seasonally variable, but annually constant, volume of water. This system was changed in the mid 1980s to a statistical formulation that incorporated interannual variation into the volumetric calculation (Light and Dineen, 1994).

Berkes and Folke (1998) argue that local communities and institutions co-evolve by trial and error at time scales in tune with the key sets of processes that structure ecosystems within which the groups are embedded. Many of the crises chronicled in Gunderson *et al.* (1995) were created by an inherent focus on one scale for management, and reformations of learning recognized the multiple scales by which the ecosystem was functioning.

Another way in which people manage for resilience in resource systems is by concentrating on sources of innovation and renewal. Many forms of catastrophic insurance provide this function, by creating a fiscal reservoir that can be tapped, should structures need to be replaced. Another mechanism that explicitly plans for renewal in resource systems is the scheme of market-based property rights systems developed for Australia. Young and McCay (1995) argue that adding flexibility and renewable structure to property rights regimes will increase resilience. They indicate that market-based property right schemes (licenses, leases, quotas or permits) should have built-in sunset (termination) to the scheme, with stable arrangements (entitlements, obligations) in the interim years. These principles complement Ostrom's (1990) findings of successful institutions allowing for stakeholders to participate in changing rules that affect them.

Finally, the ability of institutions to renew themselves following crises or to generate new and novel solutions to resource problems appears to be a pragmatic adaptation. A few key ingredients appear necessary to facilitate the movement of systems out of crisis through a reformation. In the review of management histories in Western systems (Gunderson *et al.*, 1995), these included functions

of learning, and engagement and the ability to tap into deeper understanding and trust. Lee (1993) calls this 'social learning;' it is a process that combines adaptive management within a framework of collective choice.

Weick (1995) describes a number of sources of organizational resilience including improvisation, virtual role systems, the attitude of wisdom, and norms of respectful interaction. Other authors (Berkes and Folke, 1998) describe this as cultural capital, comprised of the institutions, traditional knowledge, common property systems, which are the mechanisms by which people link to their environment. It is such linkages and connectivity across time and among people that help navigate transitions through periods of uncertainty to provide social resilience.

2.5 Summary and conclusions

Walters (1997), Johnson *et al.* (1999), Gunderson (1999b), and the cases in this volume have stressed the practical difficulties that humans face in attempting to manage ecosystems. The multiple scales of variables, cross-scale, and nonlinear interactions generate the multi-stable behaviors in ecosystem dynamics. The surprises generated by this multi-stable behavior create a range of problems for management (Carpenter *et al.*, 1999a). All of these compound the difficulties of managing, let alone managing adaptively.

Adaptive management in its early form focused on confronting the uncertainty of resource dynamics through actions designed for learning (Holling, 1978; Walters, 1986). This has evolved from a process of testing a single hypothesis about the system to sorting among multiple hypotheses, each of which may have different social and management implications (Gunderson, 1999b). Other layers of complexity arise from having adequate monitoring or data to put these hypotheses at risk (Walters, 1997). Ludwig (1995) considers harvesting strategies under increasing layers of uncertainty, and shows that increasing uncertainty generally leads to increasing caution in harvesting and a strengthened precautionary principle.

The challenges posed have a technical dimension and a social dimension. The technical challenge has two parts as well. The first is to develop a framework that will allow for a process of formulating testable hypotheses, and the second is how to choose among multiple hypotheses. The models of complex adaptive systems that appear in Carpenter *et al.* (1999a) are useful frameworks for the problem of formulating hypotheses. These have a long history of use in the process of adaptive assessment (Walters, 1997). The process of constructing these types of models is much more important than the model itself (Walters, 1986).

The technical challenge of sorting among competing hypotheses is problematic, although Walters (1986) and Hilborn and Mangel (1997) give quantitative guidance. The second challenge is the social arena. The types of organizational complexity raised by Westley (2002) and political pathologies (Pritchard and Sanderson, 2002) generate barriers for the adoption of adaptive management in Western bureaucratic agencies. Adaptive management has been socially challenged through practices such as self-serving interests of management agencies, career concerns and greed among scientific experts, and dis-information campaigns by opposing sides who exploit the uncertainty of resource issues to maintain the *status quo*.

Uncertainty pervades resource issues. That uncertainty is manifest as ecological surprises and crises. In a typology, different types of surprises have been noted: local, cross-scale surprises and true novelty. Each connotes a different type of adaptive response. Ecological crises occur when it is revealed that extant policy fails. Surprises and crises are linked to ecological properties of resilience and adaptive capacity, whereas dynamic responses are related to institutional adaptability and flexibility. This tension is in contrast to the conservative properties of resource management institutions – institutions that in many cases were created to resolve key uncertainties. Those uncertainties have technical components, political components, and stakeholder-citizen components. Few arenas exist that seem to successfully embrace these different types of uncertainties. Positive, adaptive responses appear to involve novelty – tools, models, and theory that focus on both understanding the ecological dimensions of a crisis and on the institutions that focus on creativity and learning.

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3

Nature and society through the lens of resilience: toward a human-in-ecosystem perspective

IAIN J. DAVIDSON-HUNT AND FIKRET BERKES

3.1 Introduction

There is a long history in several disciplines of trying to understand the relationship between ecological and social systems. The issue is often glossed as the nature/culture and environment/society dichotomies. Glacken (1967) has provided an extensive and wide-ranging survey of the ways in which the relationship between nature and society have been conceptualized within Western thought up to the eighteenth century. With the Age of Enlightenment, humans were extracted from the environment. The separation of nature and society became a foundational principle of Western thought and provided the organizational structure for academic departments. Since that time, Western thought has oscillated between positions in which nature and society were treated as distinct entities, and one in which articulations between the two were examined.

One of the early attempts to provide a model of natural system–society articulation was the one constructed by Karl Marx in the nineteenth century (Ingold, 1980; Wolf, 1982; Harvey, 1996). The discussion of the relationship between nature and society continued during the twentieth century in many different disciplines. There has been the human ecology of Park (1936), the cultural ecology of Julian Steward (1955), the ecological anthropology of Gregory Bateson (1973, 1979), Netting (1974, 1986, 1993), Vayda and McCay (1975), the ideas of Carl Sauer (1956) and other human geographers, the environmental history of William Cronon (1983, 1995) and Donald Worster (1977, 1988), the ethnoecology of Conklin (1957) and others (Toledo, 1992; Nazarea, 1999) and the emerging political ecology of Greenberg and Park (1994), Peet and Watts (1996) and others. The literature pertaining to the nature and society relationship spans many disciplines and has consolidated in the last two decades or so into half a dozen subdisciplines, as reviewed in Chapter 1.