Effectiveness monitoring of biodiversity in dynamic environments: Is it possible?

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Abstract

Effectiveness monitoring is undertaken to evaluate the success of management actions at achieving the goals of a particular strategy. Within the context of sustainable forest management in North America, goals for biological diversity are usually based on historic and/or current conditions with the assumption that future conditions will continually reflect the past and/or present. In practice, most monitoring is concentrated at the stand-level and focuses on cause-event relationships between indicators and current practices. However, it is usually impossible to monitor all the processes and other indicators that would enable a full cause-event explanation to be achieved; most monitoring programs ignore causality and by so doing decrease the predictability of the cause-event relationships and increase the statistical uncertainty of the results.

At the same time, the need to scale monitoring results from the stand-level to the landscape-level, through the assumption that the whole is equal to the parts, has the potential for failure over the long-term due to the dynamic nature of environments. Not only is there complexity associated with the interactive dynamics of ecosystems, but the interactions between the effects of forest management, climate change, other forms of environmental change, natural disturbance and succession mean that evaluating the success of a particular management strategy is not a simple causal deterministic problem. Consequently, a monitoring program that does not consider environmental change has the potential to acquire erroneous and confounded results which may increase the biological and economic costs of management. With so much uncertainty associated with the dynamics of ecosystems and landscapes, is this traditional methodology typically used in effectiveness monitoring going to provide the reliable outcomes and contribute to conserving biological diversity? Here, we propose that a top-down approach that considers environmental change at the landscape scale and uses a multiple-species indicator approach to evaluate the impacts of alternative management actions under the influence of environmental change could provide the means to develop a monitoring framework from which traditional effectiveness monitoring approaches can be used with increased predictability and certainty.

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Introduction

Holling (1978) has argued that monitoring should be done by all organizations involved with natural resource management, as it promotes learning, understanding, application and adjustment (i.e., adaptive management). These are all essential to the proper management of natural resources. Monitoring can be defined as the collection or creation of data for comparison to an explicit standard of performance over time (McClain 1998; Hickey 2004). In terms of the adaptive management framework, the most important approach to monitoring is termed 'effectiveness' monitoring[3] (Holling 1978; Bunnell 2003). Effectiveness monitoring involves the regular evaluation of management practices to assess their effectiveness in meeting desired management objectives (Bunnell 2003). Mulder *et al.* (1999) described the essential steps that can be used by forest managers when approaching the development of an effectiveness monitoring program, as follows:

- Specify goals and objectives;
- Characterize stressors and disturbances;
- Develop conceptual models – to outline the pathway from stressors to the ecological effects on one or more resources;
- Select indicators – to detect stressors acting on resources;
- Determine detection limits for indicators – guide to sampling design;
- Establish trigger points for management intervention; and
- Establish clear connections to the management decision process.

This paper describes some of the challenges facing forest managers as they approach effectiveness monitoring for biodiversity values in British Columbia, Canada.

Ecosystems, Causality, and the Principle of Determinism

Ecosystems are complex interacting systems composed of nonlinear interactions, but they also exist in alternating stable states over time (years to centuries) created by the influence of biotic and abiotic variables (Holling 2000). It is these alternate states of stability that Holling has identified as the source of resilience in ecological systems. In some situations, there may be multiple states for a particular system that are determined by purely intrinsic factors, such as the ash and beech forests that alternate on the same sites in southern England (Watt 1947). More often than not, extrinsic factors interact with intrinsic process to create multiple potential states.

A fundamental tenet of science is that all things are determined (Kimmins 2004). Based on this tenet it is assumed that there is an explanation for all events or conditions in the form of antecedent conditions, factors or determinants that collectively cause the event (e.g., species extinction). This has been referred to as the scientific 'Principle of Determinism'[4]and describes the importance of determinism in relation to the predictability of the event and the complexity of the determinant. Kimmins (2004) describes three forms of determinism: *causal*[5] (one antecedent), *multiple* (2 to 30 antecedents), and *statistical* (> 30 antecedents). The greater the

[3] According to Mulder *et al.* (1999) there are two other types of monitoring: '*implementation'* and *'validation'* monitoring.
[4] Described by Kimmins (2004), based on the work of Bunge (1959).
[5] Causal determinism is described as a cause-event relationship. This means that the event occurs only when the cause is present, and every time the cause is present the event will occur (Kimmins 1996)

number of antecedents, the lower the predictive ability of science because of the production of random, unpredictable results when antecedents are combined (see Figure 1).

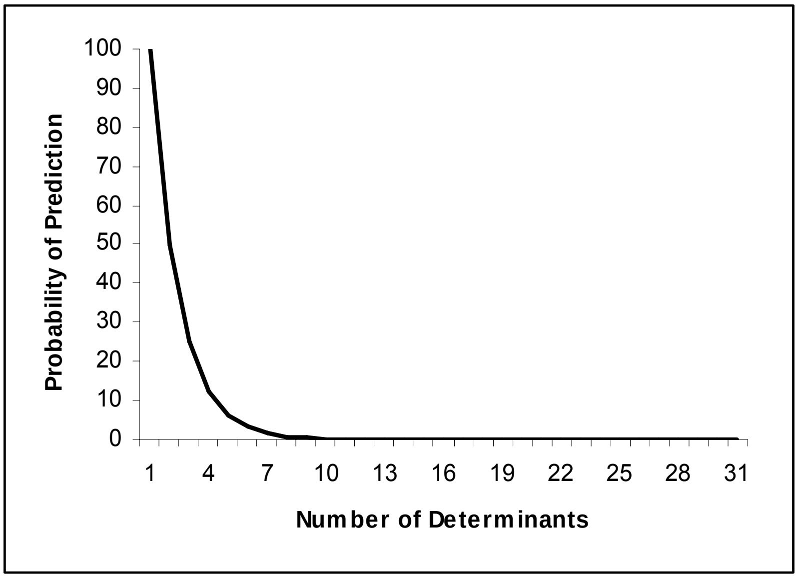

Figure 1: Predictability versus complexity through the Principle of Determinism (adapted from Kimmins 2004).

When dealing with systems that are both spatially and temporally complex, there is a need to recognise antecedents. Each time an antecedent and its causal relationship are identified, the predictive power of the study is increased. By working to increase our understanding of the antecedents associated with a particular (complex) system, the problem being examined can often be reduced to a form of determinism with higher predictive power (Kimmins 2004).

Based on a similar line of reasoning, Gunderson and Holling (2001) devised the term 'panarchy' to describe the evolving nature of complex adaptive systems. They defined panarchy to be the structure in which systems of nature are interlinked in never-ending adaptive cycles of growth, accumulation, restructuring, and renewal, that take place in nested sets of spatial and temporal scales. Figure 2 shows the adaptive cycles within an ecosystem (Gunderson and Holling 2001). Energy flows between the 'potential' of the system to accumulate resources and the 'connectedness' of the system that determines the source of variability acting on the system. At a period of low potential and low connectedness there is the possibility of resources exiting the system and pushing the system into a new state of equilibrium.

Holling (2000) and Gunderson and Holling (2001) have argued that ecosystems are not systems driven by single casual deterministic relationships, but multiple deterministic factors and influences, and this is supported by, Chorley and Kennedy (1971), O'Neill *et al*. (1986), and Kimmins (1996; 2004). Ecosystems have been identified as 'dual organisational' (O'Neill *et al.* 1986). Dual organisations are determined through the interactions of structural and functional constraints that cannot be reduced to each other in any simple way. O'Neill *et al.* (1986) found that significant ambiguities can be introduced into the understanding of systems when either functional or structural constraints[6] are ignored. Gunderson and Holling (2001) advocated that

[6] Functional constraints operate on ecosystem processes, while structural constraints operate on organisms.

by gaining an understanding of ecosystem cycles and their scales, it may be possible to identify points at which a system is capable of accepting positive change, and, from this understanding, foster resilience and sustainability within a system. Consequently, when dealing with the complexity of ecological systems, and the interactions of biodiversity within these systems, monitoring programs must reflect the interactive and non-linear effects of environmental dynamics on ecosystem processes. This will allow managers to gain an understanding of what role their management actions play in influencing biodiversity and ecosystem stability response.

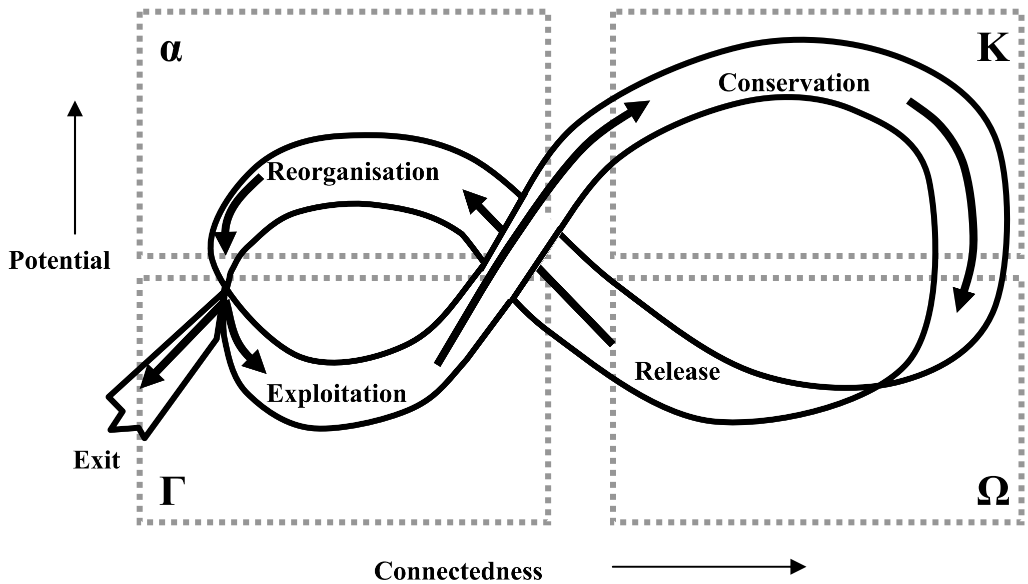

Figure 2: Holling's Figure Eight Model and a representation of the flow of events between ecosystem functions (Ω, Γ, K, α) *(*adapted from Gunderson and Holling 2001).

Monitoring Biological Diversity in British Columbia

Forest managers throughout the world generally recognize that the conservation of biological diversity values should be given a high priority. This is enshrined in a number of international agreements, such as the Montreal Process. Managers are concerned about biodiversity as damage to biodiversity values is often impossible, difficult, or very expensive to rectify (Poore 2003). It has therefore become important for certain 'indicator' species to be used to determine whether management decisions are detrimental to landscape biodiversity, or are acting congruently with conservation efforts to maintain these values. The choice of indicator species must consider not only species that represent all ecosystem types and forest structures, but also species that represent different life forms, as these will respond differently to management and environmental change. In British Columbia, the choice of indicator species is a critical aspect of most biodiversity monitoring programs (see, for example, Kremsater *et al.*2003).

There are many proposed management paradigms for selecting 'indicator' species. For example, coarse and fine filter approaches have been advocated (Noss 1987; Hunter 1990; Thompson and Angelstam 1999; Kremsater *et al.*2003; Rempel *et al.* 2004). In such cases, the coarse filter approach is holistic while the fine filter approach is species-specific. Within this context, Thompson and Angelstam (1999) have identified a grouping of potential indicator species that merit individual consideration:

- ecologically important species (keystone species)
- species sensitive to disturbance;
    - area-sensitive species (area- and edge-sensitive species);
    - species that concentrate during sometime of the year (e.g., in winter habitats);
    - rare, threatened and endangered species;
- economically important species; and
- indicator species (species that can be monitored).

Hunter (1990) defined indicator species as species that have such a narrow ecological tolerance that there presence or absence is a good indication of particular environmental conditions. This idea has been developed further by Thompson and Angelstam (1999), who suggested that indicator species can be used to evaluate the effects of change within a particular system at a variety of scales. Selection of such indicator species is difficult, given the bewildering diversity of species that could be used. However, within the context of forest management, Kremsater *et al.* (2003) have helped to resolve this issue by stating that the selection of indicator species should be guided by three broad features: sensitivity to forest practices, ease of monitoring, and usefulness of information for guiding management.

This does not mean that only one species should be selected for monitoring. Rather, as Thompson and Angelstam (1999) have stated, the choice and number of indicator species chosen should be a suite of species that represent forest change at each spatial scale of importance (stand, landscape and regional scales). This will provide a much more robust data set that can then be used to examine the impacts of particular management strategies on forest ecosystems.

While the use of indicator species is generally recognized as an important component of effectiveness monitoring, there remain a number of issues surrounding the choice of indicators before it can be certain that biodiversity values will be protected. One issue that needs to be considered is the impact of environmental change on the response of indicators to current practices. Environments are dynamic, so monitoring programs must build flexibility into their structure to account for environmental change (Innes 1998). The identification of multiple and redundant indicators based on the recommendation of indicator development studies is one approach that can provide this flexibility to deal with the issue of environmental change. This approach is congruent with Gunderson *et al*.'s (2002) recommendation to use a diversity of species that are ecological 'drivers' and a diversity of 'passenger' species[7] that could be potential drivers due to environmental change.

Dynamic Systems: The Issue of Environmental Change

To effectively monitor biodiversity in dynamic systems, managers need to evaluate the influence of environmental change, particularly climate change and climate-driven disturbance, on biodiversity in a landscape. Abiotic components that drive environmental change provide a fundamental problem for describing ecosystems in simplistic ways (O'Neill *et* al. 1986). Owing to the causality associated with these systems, it is very important for mangers to use a holistic

[7] Driver and passenger species are based on the 'Driver and Passengers' hypothesis of Walker (1992).

approach that relies on a suite of indicator species designed to represent multiple scales and functions. This holistic approach has been supported by a range of authors, including McLaren *et al*. (1998); Thompson and Angelstam (1999); Kremsater *et al.* (2003) and Rempel *et al.* (2004).

The interacting effects of environmental change can create systems with multiple deterministic relationships. In such systems, the impact of one change in the environment can influence another change through interactive and feedback effects. For example, local, regional and global changes in temperature and precipitation can influence the occurrence, timing and frequency, duration, extent, and intensity of disturbances (Dale *et al*., 2001). Dale *et al*. (2001) identified that ecosystems could resist the direct influences of climate change for decades or centuries if they were not altered by climatically-influenced disturbance regimes. Climatically-induced disturbances can lead to rapid changes in ecosystems over large areas. For example, Allen and Breshears (1998) reported a climate-induced shift in a 2,378 ha forest area over a 5 year time period (1953-1957) (Figure 3). They observed changes in the environmental variables that shifted the area from an ecosystem dominated by Ponderosa pine (*Pinus ponderosa*) forest to a Piñon (*Pinus edulis*) - Juniper (*Juniperus monosperma*) woodland. This change in the ecosystem was driven by a 5-year drought that severely stressed the Ponderosa pine, making it susceptible to infestations by bark beetles (*Dendroctonus pondersae* and *Ips* spp.). Over the following five years the beetles killed the majority of the pine, thereby reducing the forested area from ~38% to ~15% (Allen and Breshears 1998). At the end of the drought, the pine forest had transformed into a Piñon - juniper woodland that still had not shown any signs of recovery to its prior state after 30 years, due to the ability of Piñon and juniper to suppress and out compete the Ponderosa pine regeneration. Similarly, Innes (1992) argued that logging and burning had caused a major shift in the forest composition on the Taitao Peninsula of southern Chile, from a high forest dominated by *Pilgerodendron uviferum* and *Nothofagus betuloides* to a low forest dominated by *Tepualia stipularis.*. These situations provide examples of how a system can be driven into a new state of equilibrium (as outlined in Figure 2), by extrinsic factors. An interesting caveat to the Piñon –juniper study was that the response of the system was amplified by a history of fire suppression activities (Allen and Breshears 1998).

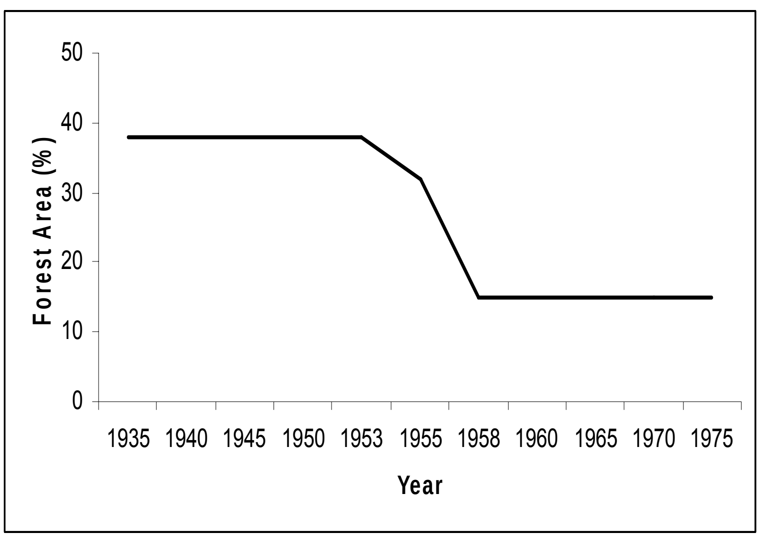

Figure 3: Climate-induced change in percent forest area between 1935 and 1975 (adapted from Allen and Breshears 1998).

If effectiveness monitoring for biodiversity values had been conducted, it could have shown a dramatic loss of some species associated with the Ponderosa pine forest, but an increase in species associated with the Piñon-juniper woodland ecosystem. In such a situation, the manager does not have the ability to mitigate or determine the cause quickly because of the interactive effects of multiple determinants.

Environmental change does not only influence forest ecosystem structure; it can also have a direct influence on the species being used as indicators. Trees, birds, amphibians and mammals show a strong interaction between the environmental variables of temperature and precipitation whereas reptiles respond significantly to temperature, but not precipitation (Currie 2001). This is supported by Hansen *et al.* (2001), who found that reptiles and amphibians are expected to increase in richness while bird and mammal richness may undergo significant reductions as a consequence of climate change (i.e., temperature and precipitation change). See Shugart (1998), National Assessment Synthesis Team (2000), and Gitay *et* al. (2002) for potential changes in temperature and precipitation as a result of climate change. The responses of indicator species to changes in environmental variables could confound monitoring results by erroneously showing an increase or decrease in biodiversity as a result of management actions. Kerr and Packer (1998) assumed that mammals will be able to track their shifting climatic zones despite anthropogenic and natural barriers and that the southern half of BC may show as much as a 10% increase in mammal diversity, with the northern half showing a possible 20% increase. If this assumption holds, then shifts in biodiversity are inevitably associated with shifts in climate, leading to many new interaction effects that could confound the results of effectiveness monitoring.

Discussion: Is It Possible?

Phenomena associated with environmental change, such as climate change and altered natural disturbance regimes, could increase the habitat for some species while at the same time reducing the habitat available for others, hence the need for a comprehensive and diverse list of species. In the current (2004) context of forest management, the goals for biodiversity conservation are being set based on historic and/or current observations. Ecological monitoring frequently relies on the statistical assumption that future observations can be modelled as a function of past observations (Bunnell 2003). Consequently, monitoring programs typically do not address causality. Since the effectiveness monitoring approach is effectively converting observations from identified cause-event relationships (e.g., a glass rod being broken by dropping it onto a stone floor (Kimmins 2004)) into assumptions (e.g., direction or rate of change (Bunnell 2003)), it cannot be expected to account for causality beyond the context of a given relationship. Causal deterministic relationships probably never occur in biological systems since any biological event involves at least two major antecedents (i.e., multiple determinism) (Kimmins 2004).

So, what are the implications of removing causality from the effectiveness monitoring programmes designed to track biodiversity values? As effectiveness monitoring generally focuses on simple causal deterministic problems, typically enveloped in multiple deterministic systems, the interaction effects of multiple antecedents on indicators are either ignored by managers, or addressed by simplifying assumptions. By using causal deterministic approaches and not recognizing the interaction effects associated with environmental change (e.g., climate change and natural disturbance), forest management may confound monitoring results by reducing the probability of predicting an effect. The probability of producing highly predictive and effective results raises questions about how managers should use effectiveness monitoring to guide the conservation of biodiversity in dynamic environments. The statistical power of monitoring to detect change in populations of an indicator species is often weak and provides too much uncertainty (Taylor and Gerrodette 1993). This is supported by Doak (1995), who showed that statistical uncertainties often exist when monitoring environmental change effects on a species. Doak found that even when monitoring data are of high quality, lag times can exist between critical levels of change and detectable changes in the indicator (e.g., grizzly bear populations (Figure 4). These findings add strength to Bunnell's (2003) observation that monitoring trends usually reflect time periods that may not provide robust results for some ecological indicators.

Two broad classes of uncertainty have been identified (Manning *et al*. 2004): statistical and structural. Statistical uncertainty is associated with parameter or observational values that are not known precisely while structural uncertainty is associated with not correctly identifying the important relationships between variables or their functional forms. By not incorporating causality, monitoring programs may face increased statistical uncertainty and may overlook structural uncertainty. Kremsater *et al*. (2003) advocated that monitoring programs must incorporate processes that are critical to monitoring long-term habitat projections for indicator species, while Doak (1995) argued that an *a priori* analysis of the potential consequences of development is required to add rigour to monitoring programs. This suggests that to monitor indicators effectively, the influences of change in the environment must first be addressed in order to reduce statistical uncertainty. Therefore, by assessing structural uncertainty, the effectiveness of the monitoring program should be increased.

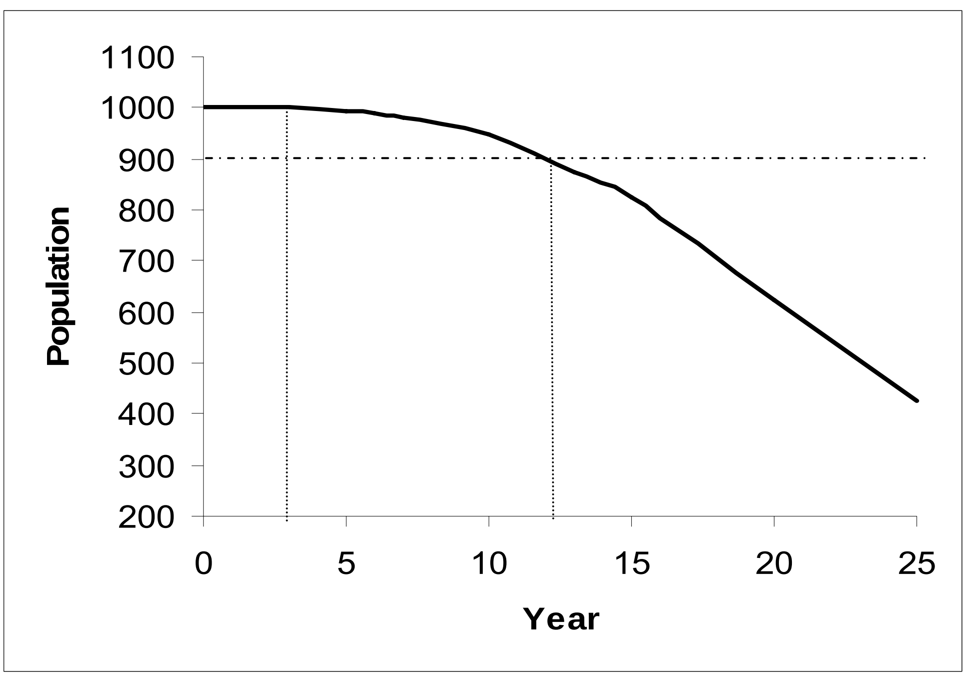

Figure 4: Change in grizzly bear (*Ursus arctos*) population in Yellowstone over a 25 year period under the influence of 1% loss of high quality habitat. 10-year marker identifies the initial point where high quality habitat falls below the point necessary to maintain a stable population and the point where a 10% decrease in population can be statistically identified (adapted from Doak 1995).

In order for effective monitoring to take place, indicators must be sensitive to stresses, respond to stresses in a predictable way, and, must be easy to measure in a cost-effective way (Bunnell et al., 2003). If managers are to consider the impacts of environmental change; they will need to gain a better understanding of how the environment will affect their chosen indicators through time and space. In this situation, how can we effectively monitor biodiversity in a cost-effective way and how do we know the extent to which indicator species will be sensitive to management induced stressors or environmental stressors[8]? These questions describe the complexity that managers and researchers face. To be cost-effective, managers often need to work at smaller scales, with indicators that are easy to measure, but these may not be the most effective for evaluating change (Innes 1998). Also, managers often need to work from the assumptions surrounding cause and event relationships. The exclusion of causality can be described as a bottom up approach where monitoring results are scaled from the effects of change at the stand-level and extrapolated to the landscape-level. One must be careful here because with this monitoring approach there is an assumption that the whole is equal to the parts and this has the potential for failure over the long-term due to the dynamic nature of environments. O'Neill and King (1998) identified that by changing scale you change the level of organisation and thus could simplify the nature and potential behaviour of the system at the landscape-scale. This means that the potential response of the indicator to stressors may change as well. Since this has implications for both the biological and economic costs of management it becomes essential that an approach that offers trade-offs between simplicity and complexity needs to be used for effectiveness monitoring.

To address this trade-off, we believe that a top-down approach needs to be used to evaluate the responses and behaviours of a wide range of species within a management area. This is supported by Chorley and Kennedy (1971) who argued that it is important to investigate the development of systems through time. Gunderson *et* al. (2002) also stated that it is important to understand dynamics of systems and their vulnerability to change. The top-down approach involves starting at the landscape scale and conducting a holistic analysis of the impacts of alternative management actions on an array of potential indicators, under the influence of stressors such as climate change and natural disturbance. Such an approach may develop and/or reinforce our understanding of the interactions between ecosystem processes and thereby provide measures of ecological resilience[9]. An understanding of this resilience will provide managers with the opportunity to successfully learn and change their management actions (Gunderson *et* al. 2002). A holistic evaluation of landscape behaviour can provide measures of ecosystem resilience under environmental change that then can be used to develop a monitoring framework from which biodiversity can be effectively monitored through the traditional bottom-up approach (Figure 5).

The top-down approach alone is not an effective approach since it can only provide an understanding of how processes influence the evolution of the system (Chorley and Kennedy 1971). The bottom-up approach allows for the step by step development of management; however, the lack of understanding of abiotic processes makes results potentially ambiguous (O'Neill *et* al. 1986). Melding the two methods into a hybrid approach allows for these limitations to be minimised and the strengths of each method to be maximised. The hybrid approach allows for increased statistical certainty associated with effectiveness monitoring results which will then translate into increased success of the adaptive management framework it drives. The hybrid

[8] For a descriptive list of management induced stressors and environmental stressors in riparian ecosystems see Kershner *et al.* (2004).
[9] Resilience allows a system to withstand the failures of management actions (Gunderson *et al*. 2002).

approach will also better enable managers to conserve biodiversity. However, due to the high degree of uncertainty associated with future events, the question of whether or not it is possible for effectiveness monitoring to conserve biodiversity in dynamic systems will remain open for some time.

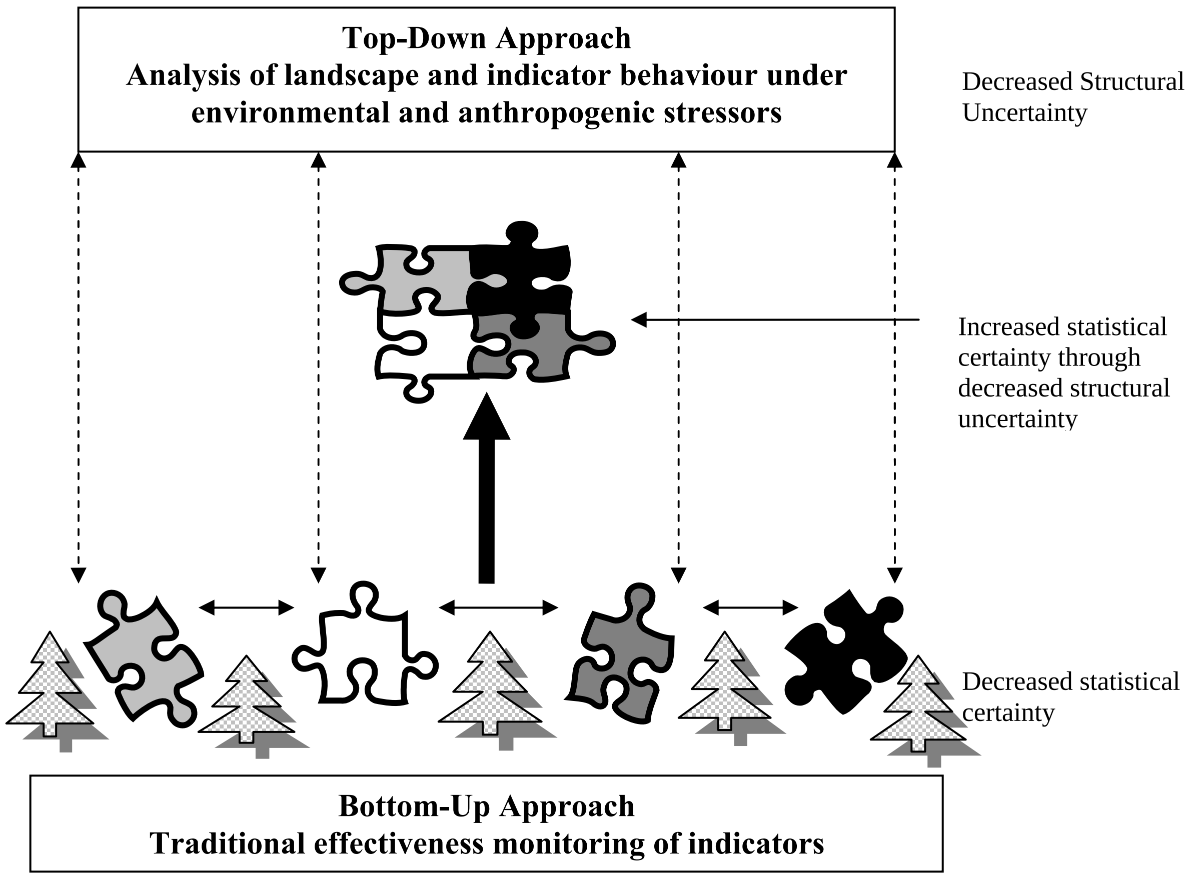

Figure 5: Hybrid Approach to developing effectiveness monitoring program for biodiversity conservation.

References

Allen, C.D. and D.D. Breshears. 1998. Drought-induced shift of a forest-woodland landscape in response to climate variation. Proceedings of the National Academy of Science. USA 95: 14839-14842.

Bunge, M. 1959. Causality: The Place of the Causal Principle in Modern Science. Cambridge, Mass.: Harvard University Press.

Bunnell, F.L. 2003. Monitoring to sustain biological diversity in British Columbia. Report prepared for Biodiversity Branch of the Ministry of Water, Land, and Air Protection. 20 March 2003.

Bunnell, F.L., B.G. Dunsworth, D.J., Huggard and L.L. Kremsater. 2003. Learning to sustain biological diversity on Weyerhaeuser's coastal tenure. The Forest Project, Weyerhaeuser, Nanaimo, BC. Online: < http://www.forestbiodiversityinbc.ca/forest_strategy/pdf/am_framework_full.pdf>.

Chorley, R.J. and B.A. Kennedy. 1971. Physical Geography: A Systems Approach. London, UK: Prentice Hall International Inc.

Currie, D.J. 2001. Projected effects of climate change on patterns of vertebrate and tree species richness in the conterminous United States. Ecosystems 4: 216-225.

Dale, V.H., L.A. Joyce, S. McNulty, R.P. Neilson, M.P. Ayres, M.D. Flannigan, P.J. Hanson, L.C. Irland, A.E. Lugo, C.J. Peterson, D. Simberloff, F.J. Swanson, B.J. Stocks and B.M. Wotton. 2001. Climate Change and Forest Disturbances. Bioscience 51 (9): 723-734

Doak, D.F. 1995. Source-sink models and the problem of habitat degradation: general models and applications to the Yellowstone grizzly. Conservation Biology 9 (6): 1370-1379.

Gitay, H., A. Suárez and R. Wilson. 2002. Climate Change and Biodiversity. Intergovernmental Panel on Climate Change Technical Paper V. Geneva, Switzerland: IPCC.

Gunderson, L.H. and C.S. Holling. 2001. Panarchy: understanding transformations in humans and natural systems. Washington, DC: Island Press.

Gunderson, L.H., C.S. Holling, L. Pritchard Jr. and G.D. Peterson. 2002. Resilience of large-scale resource systems. Pp 3-20 i*n* L.H. Gunderson and L. Pritchard Jr. (eds.). Resilience and the Behavior of Large-Scale Systems. Washington, DC: Island Press.

Hansen, A.J., R.P. Neilson, V.H. Dale, C.H. Flather, L.R. Iverson, D.J. Currie, S. Shafer, R. Cook and P.J. Bartlein. 2001. Global change in forests: responses of species, communities and biomes. Bioscience 51 (9): 765-779.

Hickey, G.M. 2004. Regulatory approaches to monitoring sustainable forest management. The International Forestry Review 6(2): 89-98.

Holling, C.S. 1978. Adaptive Environmental Assessment and Management. Chichester, NY: Wiley.

Holling, C.S. 2000. Theories for sustainable futures. Conservation Ecology 4 (2): 7

Hunter, M.L., Jr. 1990. Wildlife, Forests, and Forestry: Principles of Managing Forests for Biological Diversity. Englewood Cliffs, N.J. : Prentice-Hall.

Innes, J.L. 1992 Structure of evergreen temperate rainforest in the Taitao Peninsula, southern Chile. Journal of Biogeography 19, 555-562.

Innes, J.L. 1998. Measuring environmental change. Pp. 429-457 *in* D.L. Peterson and T. Parker (eds.). Ecological Scale: Theory and Application. New York, NY: Columbia University Press.

Kerr, J. and L. Packer. 1998. The impact of climate change on mammal diversity in Canada. Environmental Monitoring and Assessment 49: 263-270.

Kershner, J.L., M. Coles-Ritchie, E. Cowley, R.C. Henderson, K. Kratz, C. Quimby, D.L. Turner, L.C. Ulmer and M.R. Vinson. 2004. Part 1: A plan to monitor aquatic and riparian resources (PACFISH/INFISH) and biological opinions for bull trout, salmon, and steelhead. Pp 1-15 *in* J.L. Kershner, E.K. Archer, M. Coles-Ritchie, E. Cowley, R.C. Henderson, K. Kratz, C. Quimby, D.L. Turner, L.C. Ulmer and M.R. Vinson (eds.). Guide to effective monitoring of aquatic and riparian resources. General Technical Report RMRS-GTR-121. Fort Collins, CO: U.S. Department of Agriculture, Rocky Mountain Research Station.

Kimmins, J.P. 1997. Forest Ecology: A Foundation for Sustainable Management 2nd Ed. Upper Saddle River, NJ: Prentice Hall.

Kimmins, J.P. 2004. Forest Ecology: A Foundation for Sustainable Forest Management and Environmental Ethics in Forestry 3rd Ed. Upper Saddle River, NJ: Prentice Hall.

Kremsater L., F. Bunnell, D. Huggard and G. Dunsworth. 2003. Indicators to assess biological diversity: Weyerhaeuser's coastal British Columbia forest project. The Forestry Chronicle 79 (3): 590-601.

Manning, M., M. Petit, D. Easterling, J. Murphy, A. Patwardhan, H-H. Rogner, R. Swart and G. Yohe. 2004. Describing Scientific Uncertainties in Climate Change to Support Analysis of Risk and of Options. IPCC Workshop Report. National University of Ireland, Maynooth, Co. Kildare, Ireland. 11-13 May 2004.

McClain, K. 1998. A framework for monitoring indicators of sustainable forest management: First approximation. McGregor Model Forest Association, Prince George, BC, Canada.

McLaren, M.A., I.D. Thompson and J.A. Baker. 1998. Selection of vertebrate wildlife indicators for monitoring sustainable forest management in Ontario. The Forestry Chronicle 74: 241-248.

Mulder, B.S., B.R. Noon, T.A. Spies, M.G. Raphael, C.J. Palmer, A.R. Olsen, G.H. Reeves and H.H. Welsh Jr. 1999. The strategy and design of the effectiveness monitoring program for the Northwest Forest Plan. General Technical Report PNW-GTR-437. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. Portland, OR.

National Assessment Synthesis Team. 2000. Climate Change Impacts of the United States: The Potential Consequences of Climate Variability and Change. US Global Change Research Program, Washington DC.

Noss, R.F. 1987. From plant communities to landscapes in conservative inventories: a look at the Nature Conservancy (USA). Biological Conservation 41: 11-37.

O'Neill, R.V., D.L. DeAngelis, J.B. Waide and T.F.H. Allen. 1986. A Hierarchical Concept of Ecosystems. Monographs in Population Biology 23, Princeton, NJ: Princeton University Press.

O'Neill, R.V. and A.W. King. 1998. Homage to St.Michael; or why are there so many books on scale? Pp. 3-16 *in* D.L. Peterson and V.T. Baker (eds.). Ecological Scale: Theory and Applications. New York: Columbia University Press.

Poore, D. 2003. Changing Landscapes. London, UK: Earthscan Publications Ltd.

Rempel, R.S., D.W. Andison and S.J. Hannon. 2004. Guiding principles for developing an indicator and monitoring framework. The Forestry Chronicle 80 (1): 82-90.

Shugart, H.H. 1998. Terrestrial ecosystems in changing environments. Cambridge, UK: Cambridge University Press.

Taylor, B.L. and T. Gerrodette. 1993. The uses of statistical power in conservation biology: the vaquita and the Northern spotted owl. Conservation Biology 7: 389-500

Thompson, I.D. and P. Angelstam. 1999. Special Species. Pp. 434-459 *in* M.L. Hunter Jr. (ed.) Maintaining Biodiversity in Forest Ecosystems. Cambridge, UK: Cambridge University Press.

Walker, B.H. 1992. Biological diversity and ecological redundancy. Conservation Biology 6: 18-23.

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology 13: 27-73.