

# The Ineffectiveness of Effectiveness Monitoring in Sustainable Forest Management: Asking the wrong questions

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## Abstract

The role of effectiveness monitoring in an adaptive management framework for sustainable forest management planning was reviewed, with an emphasis on the need to develop clear management objectives and select appropriate baselines against which to monitor prior to implementation and effectiveness monitoring. The search for indicator species to evaluate the biological effectiveness of plans and objectives has highlighted difficulties associated with the high natural variability, biological complexity and spatially heterogeneous nature of forest ecosystems. We presented four studies from the west-central interior of British Columbia, Canada, which considered the selection of indicator species and species' response variables as the foundation for a biological effectiveness monitoring program. Research investigated riparian vertebrate species composition and abundance, ground-dwelling insect responses to dead wood availability, northern goshawk nest selection in response to logging, and the response of a weak cavity excavator to resource availability. In all cases we encountered obstacles to the effective and broadly applicable use of vertebrate and invertebrate species in effectiveness monitoring. We question the general emphasis on indicator species and species' response variables, although we recognize situations where this might be appropriate or necessary (e.g., threatened species), and support continued research on this issue through validation monitoring. We offer suggestions from current literature for improving the species approach and present an ecosystem process alternative for monitoring the biological effectiveness of forest management plans and practices.

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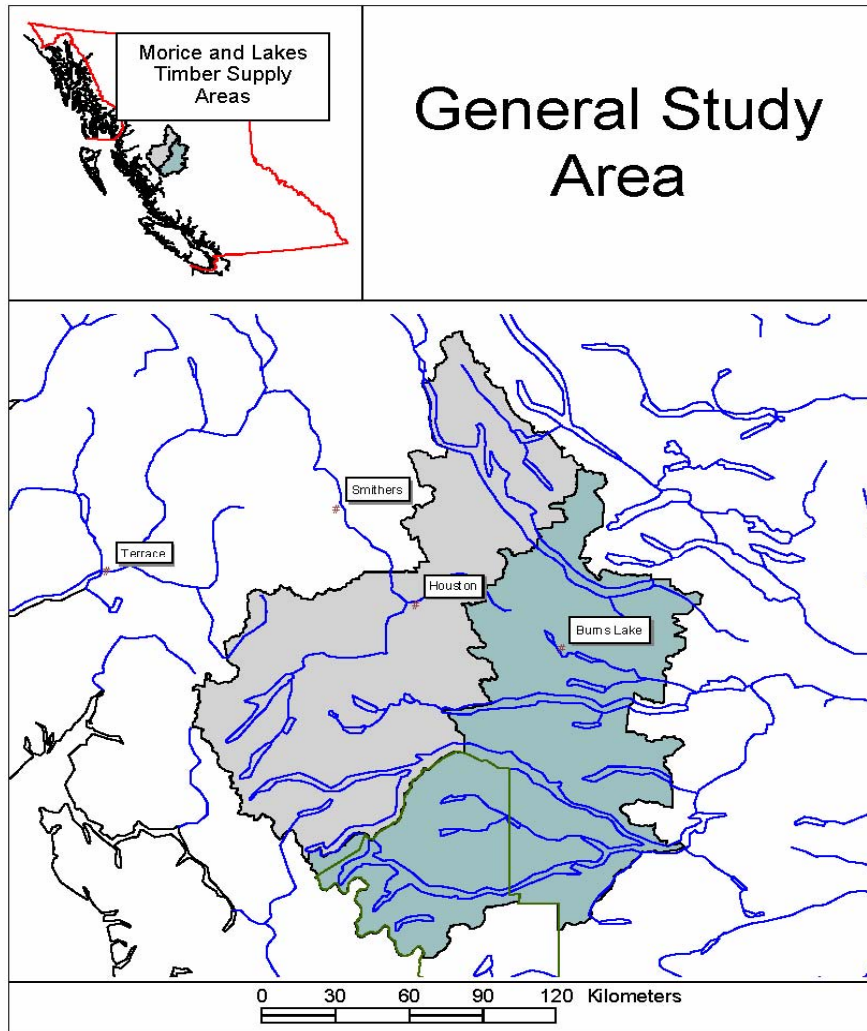
## Introduction

Monitoring the success of management objectives and strategies in achieving biodiversity conservation is a key component of the adaptive process of sustainable forest management (SFM) planning. Effectiveness monitoring is intended to determine whether a plan and its associated strategies have accomplished a resource goal, such as sustaining a viable population of a species (Franklin et al. 1999; Lindenmayer and Franklin 2002) or moving a resource condition in some desired direction (Kershner et al. 2004). The U.S. Northwest Forest Plan places effectiveness monitoring between implementation monitoring (did you do what you set out to do) and validation monitoring (establishing causal relationships between a management activity and an environmental response) (Espy and Babbitt 1994; Lindenmayer and Franklin 2002). The latter is required to establish the causal relationships on which the assumptions of what to monitor (e.g., indicators and response variables), and how to interpret indicator responses, are based. Effectiveness and validation monitoring are inextricably linked with each other and with the setting of objectives. Management objectives and strategies intended to produce predictable species' responses are often poorly developed. Effectiveness monitoring strategies should be designed concurrently with the development of management objectives and strategies as part of a clear adaptive management framework. In addition, baselines or benchmarks against which to compare monitoring results should be established *a priori*. Discussions of social (desired future condition) versus ecological (unmanaged) benchmarks are directly related to the objectives of the plan.

In the realm of SFM, land use decisions are made and management actions applied daily at a range of geographic scales; the impacts of those decisions and actions resonate across space and time. Many SFM plans are premised on lengthy lists of indicators gleaned from other published lengthy lists of indicators, with little thought as to what they are intended to represent (Failing and Gregory 2003). The selection of indicator species or species groups is often driven by values, interests or regulatory obligations (e.g., species at risk) rather than by ecological relevance. People have often focused monitoring efforts on specific taxa or communities assuming that their habitat requirements will by default maintain the ecological integrity of the entire system. This assumes direct linear causal relationships between patterns of species' responses (distribution, abundance, richness) and ecological processes. However, our understanding of the links between ecological pattern and process is limited. Non-linear causal relationships are common and patterns are often a product of processes occurring at multiple scales (Harding et al. 1998).

The complexity and enormous natural variability of ecological systems pose significant challenges to the interpretation of monitoring results (Andersen 1999; Lindenmayer and Franklin 2002). This is often overlooked as we seek to simplify the systems we have manipulated in a very simple manner (e.g., harvesting trees) (Failing and Gregory 2003). Monitoring in SFM generally avoids dealing with ecological and biological processes and focuses on measurable structural and compositional attributes (e.g., snag density or patch size distribution) and a species' or species group's measurable response to these attributes (e.g., population density or species richness).

Several cooperative projects undertaken in the west-central interior of British Columbia (BC) (Figure 1) to guide species selection and establish ecological baselines for monitoring the responses of terrestrial vertebrate and invertebrate species to forest



**Figure 1.** Studies were conducted within the west-central interior of British Columbia, Canada primarily within the Morice and Lakes Timber Supply Areas (Nadina Forest District). Chestnut-backed chickadee research was conducted northwest of Smithers.

management have highlighted many of the problems associated with current approaches to effectiveness monitoring in SFM planning. We suggest that the use of indicator species and species' response variables in effectiveness monitoring for forest management requires re-evaluation, discuss these concepts in the context of current findings, and suggest improvements and alternatives for consideration.

### **Species as Indicators**

The appropriate use of the indicator species concept requires explicit statements of what a given species is meant to indicate (e.g., other taxa, rare species, structural or compositional attributes) (Andersen 1999; Failing and Gregory 2003; Lindenmayer 1999; Lindenmayer and Cunningham 1997; Simberloff 1998). In addition, the use of indicator species is most effective when supported by a predictive understanding of responses to environmental disturbance at multiple spatial and temporal scales (Andersen 1997; Lindenmayer 1999). Researchers and

managers often use suggestions from other studies to select suitable species or species groups without any evidence of their distribution or their habitat and conservation requirements. This approach often ignores the possibility that a species may have different habitat associations in different locales within larger areas or regions; similarly so for groups of species which may be organized and/or respond differently under different site conditions (Andersen 1999). Detailed inventory and research may be required to establish species-site characteristics before selecting suitable indicator species. This can result in significant delays in the development of an effectiveness monitoring program, can make the short-term cost of implementing such monitoring prohibitive, and require long-term applied research initiatives between managers and scientists. Additional considerations for indicator selection include: adequate species abundance, distribution, and richness; mobility and detectability; threshold limits of response; appropriate spatial and temporal scales of response; and sensitivity to habitat disturbance (Andersen 1999; Brown 1997; Kremen 1992; Lavalley and Richardson 2002; Rempel et al. 2004). Two studies in west-central BC designed to identify species associated with habitats and habitat features affected by forest management illustrate the difficulties encountered whilst selecting species or species groups as indicators for effectiveness.

### **Vertebrate Species Indicators: Riparian Areas**

A study investigating the potential of individual vertebrate species or groups of species to perform as indicators of riparian ecosystem integrity was conducted in 1997 and 1998 (Todd and Mahon, in prep.; Mahon, unpubl. Data; Houston Forest Products, 2000). The study aimed to determine if riparian areas adjacent to small streams (1-5m channel width) in coniferous forests (Sub-Boreal Spruce Moist Cold Subzone (SBSmc2, Babine Variant)) (Banner et al. 1993) differed substantially from associated upland areas in either their vegetative or vertebrate species community structure, thereby enabling the identification of riparian-associated species which could be anticipated to respond in some measurable way to forest management practices. Breeding songbirds, small mammals and amphibians have territory sizes that were considered suitable for the scale of the study, were anticipated to be present in densities that would allow comparisons between habitat types (riparian versus upland), and have been commonly identified as riparian-associated, candidate indicator groups (Best et al. 1978; Bury et al. 1988; Cross 1985; Hubbard 1977).

Nine paired sites were established a minimum of 1000 m apart in mature forest stands adjacent to small streams with a minimum of 800 m of continuous, undisturbed forest cover on both sides of each stream and no other water source within 600 m of each stream (Todd and Mahon in prep). At each site, two 600 m transects were established: a riparian transect located  $\leq 5$  m from the channel bank and a parallel upland transect located 300 m upslope. Vegetation and breeding songbirds were surveyed at 4 sample points at 200 m intervals along each 600 m transect. At each sample point, geomorphic, hydrologic and ecological characteristics were recorded, vegetation was described by strata (tree, shrub, herb, and moss/lichen) within 10m radius plots, and diurnal breeding songbirds were surveyed using a variable radius point count method applied to repeat visits May through early July over two years. Small mammals and amphibians were surveyed along a 300m transect nested within each 600 m transect. A multiple species inventory approach employed live traps at 20 stations (2 traps/station at 15 m intervals), 3 pitfall arrays (4 pitfalls/array at 150 m intervals), and amphibian transect searches (Jaeger 1994) (3-100 m segments, 2 m wide), all located along the nested transect. Captured amphibians were marked and released live. All trapping occurred during the peak (July) and end (September) of the small mammal breeding season.

Trends in breeding bird and small mammal data were examined by calculating abundance, richness, percent of total abundance, percent of total richness, and percent of exclusive species for all transects by year for birds and by trapping session for small mammals. Hill's family of diversity numbers was calculated for each transect for each year and session for birds and small mammals respectively, and resemblance functions were calculated to evaluate similarity in bird and small mammal community composition and structure between riparian and upland transects (Ludwig and Reynolds 1988).

Upland areas had the same or higher bird species richness and diversity as compared to riparian areas, with abundance higher in upland transects in both years (Friedman 2-way ANOVA,  $df=1$ ,  $P<0.05$ ), (Todd and Mahon in prep.). Bird species composition and structure were similar between riparian and upland areas (range of Morisita's Index: 0.79-1.2). Species considered by others to be riparian specialists in forested habitats, such as the Northern Waterthrush (*Seiurus noveboracensis*) (LaRue et al. 1995; Machtans et al. 1996), and Wilson's Warbler (*Wilsonia pusilla*) (Machtans et al. 1996; Whitaker and Montevecchi 1999), were not limited to riparian transects ( $df=1$ ,  $P<0.05$ ). Small mammal results were inconsistent, with high variation among sites. Only diversity was significantly higher at riparian transects in both trapping sessions (Friedman 2-way ANOVA,  $df=1$ ,  $P<0.05$ ). However small mammal community species composition and structure were similar between riparian and upland transects at all sites (range of Morisita's Index: 0.80-1.00). Long-tailed Vole (*Microtus longicaudus*) and Water Shrew (*Sorex palustris*) were found in higher numbers on riparian transects ( $N=50$ ;  $N=8$  respectively) as compared to upland transects ( $N=4$ ;  $N=1$  respectively). However, capture rates were low, and these species are rare and seldom found in dense populations (Van Horne 1981; Nagorsen 1996). Vertebrate community similarities between riparian and upland sites were consistent with results indicating that vegetation composition and structure did not differ substantially between riparian and upland transects (ANOVA,  $df=1$ ,  $P>0.05$ ); although patterns of vegetation communities were highly variable at both transect types. Three of the four amphibian species known to be present in the SBSmc2 were detected, but in extremely low total numbers ( $N=9$ ), most of which were found on riparian transects ( $N=7$ ).

These findings demonstrate the difficulties encountered in selecting and sampling for suitable species or species groups as indicators in effectiveness monitoring. There were no strong riparian associations for bird or small mammal communities making the choice of what to monitor a difficult one. Bird species anticipated to be highly riparian-associated from studies elsewhere were not, illustrating the geographic and site specificity of indicator species selection (Todd and Mahon, in prep). Low capture rates of confirmed riparian-associated species for this area (small mammals and amphibians) highlight the need for trapping continuously over long time periods in order to detect response trends. Overall, there was a high variability between sites, sessions and years, confounding results and demonstrating the need for both longer term sampling and increased replicates for baseline establishment and monitoring. Costs associated with such intensive monitoring efforts may limit their applicability in SFM.

### **Invertebrate Species Indicators: Woody Debris**

A recent study investigated the potential of Coarse Woody Debris (CWD)-associated ant species to act as indicators for monitoring the effectiveness of CWD retention practices in forest management (Higgins and Lindgren 2005). Ants (Hymenoptera: Formicidae) were considered candidates for monitoring as they are generally abundant, ubiquitous, relatively cost-effective and easy to sample, and were believed to be tightly associated with CWD in cooler boreal and

sub-boreal climates, selecting nesting habitats such as dead wood which maximize daily heat gain (Higgins and Lindgren 2005). Further, ants are known to contribute significantly to a range of ecosystem and biological processes such as soil nutrient turnover, seed dispersal and predation (Higgins and Lindgren 2005; Hölldobler and Wilson 1990).

The presence, abundance and diversity of CWD-associated ants in stands of varying ages regenerating after harvesting and undisturbed late-seral stands in the SBSmc2 were assessed (R. Higgins, pers. comm., Morice-Lakes Innovative Forest Practices Agreement 2004). Pine (*Pinus contorta* var. *latifolia*) leading site-types investigated included: unmanaged late seral stands and harvested (clear cut) stands at seral ages 2-3, 8-10, 13-15, and 23-25 years post harvest. Sampling was applied at 2-4x100m strip plots in each of three 1 hectare plot replicates of each stand type (n=6). Replicate 50 m<sup>2</sup> sub-samples (maximum per stand n=48) derived from strip plots were used to sample CWD for associated ant fauna and physical attributes. Sampling was positioned at least 50m away from an edge, and CWD-associated ants were sampled by destructive searching of logs and by pitfall traps, environmental attributes such as temperature were measured, and CWD attribute data (e.g., volume, piece size) were collected using the sampling methods detailed in Higgins and Lindgren (2005).

Of 11 species collected, nine were found nesting within CWD. A seral response pattern was observed for CWD-nesting ants (sub-samples lacking ant fauna were excluded; sample sizes varied by stand type: unharvested (n=14), harvested 2-3 (n=18), 8-10 (n=31), 13-15 (n=48), and 23-25 (n=14)). A physically small ant species with small colony sizes, *Leptothorax canadensis*, dominated the ant fauna in early seral stands but steadily declined with time since harvest (R. Higgins, unpubl. data). Large ant species which form large nests, such as the large red wood ant (*Formica aserva*) and the boreal carpenter ant (*Camponotus herculeanus*) were not well established in the younger stands, peaked in 13-15 and 23-25 year old post-harvest stands respectively, and then rapidly declined as stands aged. Overall ant abundance peaked in the 13-15 year old post-harvest stands (N=488 colonies; more than twice the number counted in the most immediate earlier and later seral stages). As expected, the mean overall temperature of soil litter in harvested stands was significantly higher (P<0.05) than in non-harvested stands (Higgins and Lindgren 2005). In general, CWD-associated ants were largely absent from late seral unmanaged stands in SBSmc2 (3% of CWD pieces (n=333) hosted ant colonies), while 49% of CWD pieces (n=739) in 8-10 year old harvested stands (Higgins and Lindgren 2005) and 88% of CWD pieces (n=553) in 13-15 year old stands (R. Higgins, unpubl. data) hosted colonies. While CWD volumes did not differ significantly between 8-10 year old harvested and unharvested sites (P=0.1082), unharvested sites had significantly (P<0.0001) larger piece sizes, with harvested sites showing a high degree of physical damage (i.e. crushed wood and splintering) (Higgins and Lindgren 2005).

Results indicated that the responses of CWD-associated ant species to canopy removal and CWD retention levels are temporally specific. Late seral stands sampled in this study contained an extremely depauperate CWD-associated ant fauna despite the occurrence of abundant CWD-habitat. It appears that canopy re-establishment creates cool conditions intolerable for most CWD-associated ants (Higgins and Lindgren 2005). This illustrates the importance of selecting the appropriate baseline conditions for monitoring, as it was clear that late-seral stands were an unsuitable reference against which to compare CWD-associated ant species' responses to retention practices. Further investigation is required into ant-CWD relationships in naturally disturbed stands (e.g., burns) to develop meaningful baselines for monitoring ant responses through early seral stages of regeneration after harvesting. CWD is a dynamic resource, and managing for it requires objectives and monitoring premised on an understanding of persistence and recruitment through time. It would appear that CWD-associated ants provide

no indication of the condition of mid- and late-successional CWD and related processes on which other CWD-associated taxa may depend. Observed responses in this study may be individualistic to ants; knowledge of the dynamics of only a few species may not be reflective of general ecological change or other invertebrate taxa (Andersen 1999; Fager 1968).

## **Response Variables**

Species' response variables can range in scale and complexity from those applicable to the population or community (e.g. density, abundance, species richness, diversity) to more complex responses at the level of the individual (e.g. territory size, resource selection, fitness measures such as reproductive success and survival). Given the often non-linear and multi-causal nature of response, risks of type II errors loom (i.e. not finding a response when one is present) and accurate interpretation of an observed response may be obscured by our limited understanding of a species' ecology. Habitat supply models are often used to simplify systems based on observations of structural and compositional habitat variability, limited understanding of species-habitat relationships, and predicted species' responses to habitat change. Recently, population viability models have been linked to habitat supply models to attempt to factor in species' responses to dynamic population parameters. Effectiveness monitoring is often aimed at determining if responses to management objectives predicted by model outputs are in fact occurring.

The multi-scalar aspects of species' responses are often overlooked and monitoring outputs may be inappropriately interpreted relative to the scale at which the data were collected. Local species' responses can be influenced by features of the surrounding landscape matrix such as seral stage distribution, patterns of land-use and patch size (McGarigal and McComb 1995; Stauffer and Best 1980; Saab 1999; Shirley 2004; Wiens et al. 1993). We often monitor species' responses to structural attributes defined in the management application when species may be responding to cues we are unable to perceive, to other characteristics that are more difficult to measure, or to functional interactions between attributes (Freemark and Merriam 1986; Nilsson et al. 1988; Shirley 2004; Young 1996). Monitoring may indicate no effect from management when in fact the effects are either masked or will develop over time (lag). Species' responses can functionally lag or be confounded due to biological processes such as predation, intra- and inter-specific competition for resources, limited dispersal capability or extreme mobility, and intrinsic population processes such as territoriality, dispersal and cycling. A number of related issues were encountered during two studies in west-central BC of species' associated with habitats and habitat attributes of late seral forests and their responses to forest management practices.

## **Northern Goshawk Nest Area Selection**

A multi-year study to determine the effects of timber harvesting on reproductive success within Northern Goshawk (*Accipiter gentilis*) nest areas began in 1995 in the Interior Cedar Hemlock (ICH) and Coastal Western Hemlock (CWH) biogeoclimatic zones (Banner et al. 1993) and extended east into the Sub-boreal Spruce (SBS)(Mahon and Doyle 2005). Long-term data in Europe, and limited data in North America, have demonstrated a correlation between population declines and extent of forest development (Crocker-Bedford 1990; Widen 1997). Impacts to nest areas were suggested as the major factor in the observed response (Crocker-Bedford 1990) as goshawks exhibit strong nest area fidelity and there is a high overlap in resource selection by goshawks and foresters.

Adaptive management harvesting trials were established in a formal experimental framework to assess the impacts of logging within nest areas (24 ha) on reproductive success (Mahon and Doyle 2005). Of the 79 nest areas located in the ICH/CWH and SBS, treatments included a range of harvest levels (n=27 nest areas) and controls (n=52 nest areas). Nest area annual reoccupation and nest productivity (the number of fledglings produced per nesting attempt) were the response variables monitored annually from 1996 to 2002. Long term monitoring was considered key to addressing any potential lag in response. There was no difference in reoccupation rates of nest areas between treatments and controls (P=0.89). The mean number of chicks fledged per nesting attempt did not differ between treatments and controls (P=0.77). Even for treatments areas with > 50% of the nest area harvested (N=7), reoccupation rates and nest productivity were not lower than at controls (Mahon and Doyle 2005).

For goshawks in this study, there were no apparent simple relationships between habitat quality and availability, habitat selection and reproductive success at the nest area scale. Even when one of the best response variables reflective of population fitness (reproductive success) is monitored, the response may not exhibit a direct relationship to habitat quality. In this case, fidelity to nest areas may override response to reduced suitability from logging, and result in a lag effect before goshawks relocate to more suitable habitat (Mahon and Doyle 2005). The results of Mahon and Doyle (2005) are consistent with another recent study in Europe (Penteriani and Faivre 2001). If longer-term results continue to show no decreased reproductive success by goshawks at nest areas modified by timber harvesting, this may support the theory that habitat changes at the foraging area scale are the primary factor affecting goshawk populations (Widen 1997). To test this will require expansion of studies to multiple scales (territory and landscape) and the examination of community level relationships and biological processes (e.g. predator-prey). Although occurring at low densities, highly mobile, and difficult to observe, Northern Goshawks are a good indicator candidate as they strongly select for mature forest, are linked to spatially-fixed areas (high nest area fidelity), and operate at scales relative to forest management. However, we have not been able to define clear response variables, raising the question of using species response variables in effectiveness monitoring if we cannot yet sort out the situation for such an obvious mature-forest dependent species as goshawk.

### **Chestnut-backed Chickadee Habitat Selection**

A three year study of the breeding ecology of Chestnut-backed Chickadees (CBCH) was undertaken from 2001 to 2003 to examine nest survival and habitat selection across a range of partial cut harvest levels in the ICH (Mahon and Martin 2006). CBCH are obligate secondary cavity nesters and thought to be directly dependent on the density of suitable cavities in wildlife trees (live and snag). As such, nest success was predicted to be higher in stands with a higher availability of suitable nest sites and CBCH were anticipated to be good indicators for monitoring the biological effectiveness of retention strategies aimed at conserving wildlife trees. The influence of year, habitat and predators (density and proximity) on nest survival was examined by monitoring 69 natural cavity nests. Multiple scales were evaluated: the stand (three uncut and five partial cut 19-24 ha stands), the nest patch (0.03 ha patch centred on nest trees), and the nest tree. Breeding density was assessed and nest sites monitored to determine nest fate (successful or failed). Habitat variables, habitat availability and habitat use were described at the three scales. Logistic regression was used to predict nest outcome and an information-theoretic approach was used to determine support for models representing alternative hypotheses for differences in nest survival time due to year, habitat and predator effects.



The only variables which predicted nest survival time were year ( $P=0.009$ ) and nest height above ground ( $P=0.03$ ) (Mahon and Martin 2006). No relationship was found between availability of resources (tree, dead tree, and diseased tree density) and the average CBCH breeding density and persistent use of territories (all  $P$  values  $> 0.05$ ). Most of the habitat variables examined did not influence nest survival. Partial cutting did not appear to negatively impact the availability of suitable nest sites for CBCH pairs. Dramatic annual variation in CBCH nest failure, ranging from 24% to 82%, was unrelated to harvest treatment. Further, predator mortality rates showed strong annual variation, with  $>57\%$  of nests depredated in 2000 and 2001, while  $<15\%$  were depredated in 2002 and 2003. Red squirrels (*Tamiasciurus hudsonicus*) were the main nest predator and annual changes in food availability for red squirrels may be the cause of CBCH nest failure.

Results suggest that CBCH habitat use is limited by territorial behaviour and not availability of resources, and the fitness of the birds may have more to do with temporal variation in the habitat quality of their main predator than with the availability of suitable habitat elements or a change in habitat use by CBCH (Mahon and Martin 2006). This illustrates the difficulty in teasing out a response to habitat change buried with the environmental complexity of inter- and intra-specific competition and predation. We have a very poor understanding of how interactions between species, their predators, and their environment (habitat characteristics, food availability) change as a result of habitat manipulation or temporal variation in influential variables (e.g., prey cycles, cone cycles, resource pulses). Understanding these interactions is critical; without it our ability to interpret response variables and therefore use these species as effective indicators is restricted.

## Improvements and Alternatives

A review of these studies from one geographic locale within BC reveals significant obstacles and pitfalls to the use of species in effectiveness monitoring. Effectiveness monitoring is frequently directed at focal species (e.g., indicators, keystones, umbrellas) and our limited resources are committed to finding species or species groups and suitable response variables to monitor effectiveness of conservation efforts (e.g. forest management prescriptions). However, current approaches to effectiveness monitoring are proving ineffective, certainly in the short-term. Results from our studies have yet to yield workable indicator species or response variables for monitoring the effectiveness of our management practices (e.g., riparian management, CWD and wildlife tree retention, and mature forest retention patterns). Are we, therefore, asking the right questions?

1) Can a species or species group act as an effective indicator for some other value or group of values? Our studies and others indicate that to do so with confidence requires a correlative understanding of relationships between indicators and the other values they are intended to represent. It also requires a predictive understanding of responses to environmental disturbance and the ability to set meaningful baselines against which to monitor. And it requires the resources to sample with sufficient intensity over sufficiently long time periods to both establish the appropriate indicators and potentially detect a response.

2) Will the species or species group respond to management in some measurable and detectable way? Again, to do so with any confidence requires a detailed understanding of species autecology and causal relationships at a range of scales. These are fairly daunting prerequisites entailing a significant investment of time and resources to develop the understanding necessary to create a reliable and informative species-level effectiveness monitoring program.

There are ways to improve our ability to interpret the results from species-level monitoring. We can start improving the indicator selection process by asking better decision-focused questions, applying better selection criteria and developing better decision making tools (Andersen 1999; Failing and Gregory 2003; Lindenmayer 1999). Selection of species based on prior study must be done in the context of local conditions and local knowledge. We can improve the ability to deal with uncertainty and complexity through better study designs, improved statistical rigour and analysis, and the use of non-classical statistics (Bennett and Adams 2004). We can also choose more ecologically meaningful species and response variables that attempt to grapple with ecosystem complexity and heterogeneity. Soule et al. (2003) have suggested that management and monitoring should focus on the ecologically effective densities (densities that maintain critical interactions) of highly interactive species whose absence leads to significant changes in their ecosystems and may include, but are not limited to, keystone species. Maintaining and monitoring such ecological interactions will address the non-linear and often highly directional functional relationships found in trophic cascades and other resource flows (e.g., Martin et al. 2004). Carey (2000; 2003) emphasizes the importance of 'biocomplexity' in ecosystems and landscapes as a function of the spatial heterogeneity of habitat elements, and concludes that change in the structure of biotic communities (e.g., trophic pathways and food webs) in response to changes in heterogeneity due to forest management is more informative than the response of a single species. Unfortunately, species-level monitoring approaches revised to encompass complexity and interactivity – while more ecologically robust – continue to be limited in their global applicability and the need for a considerable investment of resources to promote the understanding of interactive species, trophic relationships, and community and species' ecologies.

Clear-cut harvesting does not produce the same pattern of complexity thought to underlie the recovery of ecosystems following large scale disturbance (Franklin et al. 1995; Perry and Amaranthus 1997). Spatial and structural heterogeneity and habitat complexity produce more biologically diverse ecosystems more resistant to negative impacts (Carey 2003; Kershner 2004). Recent planning initiatives in the study area (Figure 1) have focused on the influence of spatial complexity and heterogeneity on biodiversity conservation (Todd, unpubl. report; Morice Land and Resource Management Plan 2004; Todd, in prep.). If we begin to set management objectives to address complexity and heterogeneity at a range of scales (e.g., partial cutting in stands and dynamic reserve designs in landscapes), then effectiveness monitoring will need to keep pace.

The expansion of species-level monitoring to encompass biological complexity and interactivity is a possibility but is laden with obstacles. Many studies have looked at monitoring structural and compositional changes in response to management, with inferred ecological implications related to simplification and loss of structural heterogeneity. However, these authors fail to find consistently predictable and/or globally applicable responses of individual or combinations of structural attributes to harvesting (Bradbury 2004, DeLong et al. 2003, McCleary and Mowat 2003, Sullivan et al. 2001). Issues of variability and site specificity continue to prevail. Though many studies continue to restrict investigations to early versus late seral, cut versus uncut comparisons (Bradbury 2004; Sullivan et al. 2001), others have attempted to address temporal variability and complexity through retrospective examination of structural classes (McCleary and Mowat 2003) or chronosequences (DeLong et al. 2003).

To address effective monitoring of management within the context of spatially and temporally dynamic, heterogeneous and highly variable forest ecosystems, we suggest we look to the aquatic and hydrologic literature for clues. Inherently dynamic, interactive and highly variable,

aquatic systems have required monitoring approaches that are process oriented. Although limited in the past by their own versions of oversimplification (e.g., flow monitoring considered in isolation from other geomorphologic or successional factors), hydrologic effectiveness monitoring has focused on the direction and rates of change in habitats and other system parameters over time as a function of management (Bunn et al. 1999; Kershner et al. 2004; Poff et al. 1997; Robinson et al. 2002; Warfe in prep.). Ecosystem processes are expressed through rates of change; the associated effects of anthropogenic stressors on rates and direction of change are evaluated (Kershner 2004).

In forest ecosystem management we should consider following suit; expand our view out to the ecosystem processes themselves rather than restricting ourselves to look for responses in pattern. Biological complexity and spatial heterogeneity and their associated patterns of species' responses are maintained by functional ecosystem processes acting on the structure and composition of forests, affected by disturbance processes like fire and land use. Developing ecosystem process variables as functions of rate and direction of change would embrace the spatial and temporal dynamic variability of forest ecosystems. Biotic ecosystem processes such as recruitment, growth, mortality, and decomposition/decay are ecological constants. Abiotic processes are affected by geomorphologic and climatic variables as well as biotic influences (e.g., carbon and nitrogen cycles). The rates at which ecological processes proceed, the directions these processes proceed in, and the scales at which they manifest themselves (pattern) are not constant, but likely to be strongly correlated to the biotic and abiotic conditions on which and in which they proceed post-disturbance and are therefore sensitive to the effects of forest management on environmental conditions.

Ecosystem process variables relevant at a range of scales and levels of biological complexity may be cost-effectively derived from a wide range of parameters we already measure or collect data for including vegetative growth rates and productivity, survival rates, mortality rates, rates of fall down, decay rates, and fire return intervals. Suggestions from the aquatic monitoring literature focus on the hierarchical organization of systems by scale, level of organization (heterogeneity and complexity) and ecological process (Kershner 2004; Warfe in prep).

Ultimately, we are going to want to interpret process responses in terms of floral and faunal species' responses as we have SFM plans that contain objectives for the maintenance of biodiversity or viable populations of threatened species. As Lindenmayer (1999) aptly summarized, the study of indicator species and species' responses will contribute valuable new information about forest ecosystems even if the particular species targeted for study does not prove to be a reliable indicator of other taxa. However, we contend that the scientific exploration of relationships of pattern (species' responses) to ecological process could remain within the realm of validation monitoring, with effectiveness monitoring focused on the responses of ecosystem process variables to forest management.

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