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A new approach to forest biodiversity monitoring in Canada

Stan Boutin [a,*], Diane L. Haughland [a], Jim Schieck [a,b,c], Jim Herbers [b,c], Erin Bayne [a]

[a] Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada T6G 2E9
[b] Alberta Biodiversity Monitoring Institute, Alberta Research Council, PO Bag 4000, Vegreville, Alberta, Canada T9C 1T4
[c] Alberta Biodiversity Monitoring Institute, University of Alberta, Edmonton, Alberta, Canada T6G 2E9

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ABSTRACT

Society increasingly asks forestry agencies to consider maintenance of biodiversity alongside tree growth and wood supply when managing forests. In this paper we explore the mechanisms that managers currently have available to meet this expectation. When assessing three key indicators (trends in ecosystem and habitat extent, trends in distribution and abundance of species, and changes in status of threatened species), we found that managers lack the information for consistent, scientifically rigorous, unbiased reporting on the impact of their management on biodiversity in Canada. We outline the key characteristics of a controversial biodiversity monitoring approach that is taxonomically broad and cumulative-effects oriented. We argue that programs designed specifically to monitor biodiversity, although sometimes criticized as inefficient and ineffective, are statistically and biologically robust in the long term, especially compared to the status quo. Given the nature of biodiversity and the diversity of human impacts, if forest management agencies are sincere in their desire to manage biodiversity, they need to devote the same effort and scientific rigor used to monitor tree growth and harvest rates to develop standardized protocols and rigorous sampling designs for biodiversity monitoring. Federal and provincial governments, as well as the scientific community, will need to cooperate with and support forest managers in this new endeavour.

1. Introduction

"You manage what you measure" is an axiom in public and corporate governance (Lowenstein, 1996) and forest management is no different. Consider the time, energy, and money devoted to the measurement of forest inventory, regeneration, and growth: Canada and the United States together spent an estimated $76.7 million through their national inventory agencies in 2008 (Natural Resources Canada, 2007; United States Department of Agriculture, 2009). These parameters are the core of wood supply calculations and performance of the industry is judged against well-defined targets and goals (e.g., Sharma et al., 2008). In the last 20 years societal pressure has prompted forest managers to consider a wider array of ecological goods and services (Hunter, 1990; Carey and Curtis, 1996; Gilmore, 1997), many of which fall under the umbrella of biodiversity. The forest industry and research community have responded by developing and investigating new stand- and landscape-level practices designed to promote biodiversity while maintaining wood supply within the broader ecosystem management approach to forest management (Angelstam and Pettersson, 1997; Kohm and Franklin, 1997; Bergeron et al., 1999). Biodiversity maintenance is frequently listed as a key criterion in forest management plans and forest certification schemes (Canadian Council of Forest Ministers, 2006; Montréal Process, 2007; Canadian Standards Association, 2008). It is clear then, that biodiversity is something that society expects industry and government to manage, and to do so, we need to measure and report on it. In fact, the degree to which this is achieved may be a strong indicator of just how little or how far the forest industry has progressed in its commitment to moving beyond the singular focus of wood supply to the broader objectives incumbent in ecosystem management.

In this paper, we explore how biodiversity is measured in forests managed for timber production in Canada with the intent that this fundamental step is necessary to assess the success or failure of the broad array of new management practices that are being developed and instituted to enhance or maintain biodiversity. We wished to determine how well we could answer the following question: How is biodiversity changing on forests managed for long-term wood production? Resources that do not have an accepted measurement system are often marginalized in decision-making processes and, understandably, there is little accountability if they are not managed sustainably (Norton, 1998). It is our conclusion that current approaches to biodiversity

* Corresponding author.
E-mail address: stan.boutin@ualberta.ca (S. Boutin).

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monitoring do not meet the needs of forest managers or society because of a mismatch in objectives between current and desired monitoring programs, the limited temporal and geographic scale of monitoring, the limited scope of biodiversity being measured, and the sporadic reporting of results. While most of these issues are not new (Lindenmayer, 1999; Noon, 2003; Green et al., 2005; Lengyel et al., 2008), the solution we advocate is controversial.

2. Biodiversity: what is it?

Although the term 'biodiversity' is familiar to most of the public, what it represents to each person varies tremendously. Magurran (2004) and Maclaurin and Sterelny (2008) provide a description of the complex evolution of the concept but in short, the origin of the term biodiversity relates to concern over human-caused loss of species on the planet. However, biodiversity has now become a catch-all for everything from variation in genes, populations, communities, ecosystems, and landscapes to "healthy" functioning ecosystems and the goods and services they supply. There is the added confusion that some value biodiversity (variation in genes, species, etc.) in its own right, separate from its utility to humans while others view biodiversity simply as a surrogate of what society really wishes to conserve, which is ecological function and associated ecological goods and services (Maclaurin and Sterelny, 2008). The fact that biodiversity has now become a generic term creates the first major problem for biodiversity monitoring; almost any ecological variable could be considered as either an indicator or component of biodiversity and conversely, it will never be possible to measure all that is biodiversity.

The only tractable solution for biodiversity monitoring is to select surrogates; measures that serve as a "thermometer" (Maclaurin and Sterelny, 2008) for the status of biodiversity as a whole. Not surprisingly, there has been endless and ongoing debate over which surrogates are best suited for the job. Various individuals have attempted to provide some structure to this dilemma (particularly for forestry—e.g., Noss, 1990; Lambeck, 1997; Hagan and Whitman, 2006) but no consensus has been reached. The Convention on Biological Diversity (www.cbd.int) has integrated many of these suggestions into a working list of biodiversity indicators to track our progress toward reducing biodiversity loss. Under "Status and Trends of Components of Biological Diversity" (United Nations Environment Programme, 2009) they include:

1. trends in extent of selected biomes, ecosystems, and habitats,
2. change in the status of threatened species and
3. trends in abundance and distribution of selected species.

For the purposes of this exercise, we accept these three indicators as a reasonable place to begin and we will assess how we measure these indicators of biodiversity in the forested regions of Canada. The CBD suggest other criteria, including the coverage of protected areas (United Nations Environment Programme, 2009). We concur that this is an important indicator but do not consider it further because measuring and reporting on this indicator is not controversial or particularly difficult or costly to implement.

We are interested in monitoring and reporting of biodiversity at the scale at which forest harvest planning is done; i.e., the temporal and spatial scales that harvesting and silvicultural practices are planned so as to maintain long-term wood supply. This is on the order of millions of hectares and over multiple harvest rotations. Management is often the responsibility of individual companies that are the primary operator within a designated area. Following convention, we refer to these as forest management areas (FMAs). However, we also believe it is important to compare biodiversity across FMAs to have the capability of comparing different FMA-based ecosystem management approaches and reporting at provincial and federal scales. We will assess how well biodiversity indicators, as measured by forest managers, serve as surrogates for biodiversity as a whole, the scientific quality of information used, and the spatial and temporal scale of measurement and reporting.

3. Current forest biodiversity monitoring efforts in Canada

3.1. Trends in selected biomes, ecosystems, and habitats

We begin by assessing the measurement of trends in biomes, ecosystems, and habitats. While relevant for global reporting, biomes are too broad for FMA monitoring so we will not consider them further. Identifying and mapping selected ecosystems such as forests, grasslands, wetlands, and peatlands is becoming increasingly common using remote sensing, and the potential for greater detail is growing with each technological advance (Wulder et al., 2008a,b). At present, however, Canada lacks both national and provincial long-term monitoring programs to estimate trends in ecosystem extent. Estimates of forest extent from consecutive national forest inventories are not comparable due to changes in methods (Canadian Council of Forest Ministers, 2006) and the lack of standardized definitions of ecosystem types. As a result, at the scale of an FMA, most forest companies report on how the productive forest land base is altered by their activities and they may summarize the area occupied by each ecosystem type (definition and number reported varies by FMA) using an inventory derived from manual interpretation of aerial photographs, with relatively few resources dedicated to ground-truthing through field plots (Leckie and Gillis, 1995; Gillis et al., 2005).

The lack of standardized definitions and types is exacerbated when it comes to habitat monitoring. Tree canopy species, tree height (age), and tree density are the prime elements used to designate stands, the fundamental management unit for forest harvesting and silviculture. This is normally done by manual interpretation of aerial photography managed by either provincial authorities or FMA holders to create a complete inventory. In either case, photo-interpretation protocols are standardized and some ground-truthing is done. Stand types are then re-assigned to habitat types but the names given to these habitats and the criteria for how stand inventory variables are used to define each habitat is left to each FMA holder. There are no standardized protocols and the rationale for how habitats are defined is highly variable. Focal species such as exploited species (e.g. ungulates), rare or threatened species (e.g. grizzly bears (*Ursus arctos*)), or species requiring attributes thought to be negatively affected by forest harvesting (e.g. woodpeckers (*Picoides* spp.) requiring large snags) are often important drivers. This method of habitat designation works well if the habitat requirements of the focal species are well known (a small proportion of all species living in forests) and the habitats can be identified via forest inventory attributes (tree species, height, and density). It is far more difficult, if not impossible to track habitats or elements such as shrub cover, snags, and downed wood which are important to many forest species (Jonsson and Kruys, 2001).

Once habitats are defined, companies track changes on their FMA by updating the forest stand inventory yearly to include harvesting and large-scale disturbances such as fires or insect outbreaks. The large-scale disturbances are monitored and mapped by government agencies (e.g., Alberta Sustainable Resource Development, 2007). Aerial photography and interpretation is done at infrequent intervals (10–20 years, Leckie and Gillis, 1995) although reporting is usually done more frequently. Given that there is no standardization of habitat definition, it is difficult to summarize beyond the scale of the FMA. Even within FMAs there is the strong potential for "creeping baselines" as conditions, personnel and human perceptions change over the long term (e.g., Sáenz-Arroyo et al., 2005). In conclusion, in contrast to substantial monitoring of trees and productive forests, forest companies assess trends in a limited number of habitat types and the lack of standardization of these habitats across FMAs makes it difficult to report at provincial and federal levels.

3.2. Changes in the status of threatened species

Provincial and federal agencies, through the Committee on the Status of Endangered Wildlife in Canada (COSEWIC, www.cosewic.gc.ca) have an ongoing program of assessing the status of species considered to be under some risk of extinction. The information and methodology used are similar to the International Union for Conservation of Nature (IUCN, www.iucn.org) red list protocols which rely on criteria related to population size and changes in distribution and abundance (Rodrigues et al., 2006). The information available to assess species is highly variable ranging from scientifically robust, rigorous population size or trend estimates (e.g. grizzly bears, woodland caribou (*Rangifer tarandus*) to citizen-based records of occurrence (Shank, 1999). In addition, the effort and scientific rigor devoted to monitoring of threatened species is highly variable both over space and time. Species in high risk categories (threatened or endangered) receive the most attention because these designations trigger specific management actions by the Canadian Species at Risk Act (SARA, Government of Canada, 2002). Forest companies tend to list the number of threatened or endangered species considered present on their FMA but it is rare to have well-designed scientifically rigorous programs to monitor changes in status of these species. In many cases, habitats used by a species at risk are monitored rather than actual measures of population size and distribution; the latter is normally left to provincial or federal agencies.

The IUCN Red List Index (RLI) was developed to measure trends in overall risk of extinction for selected taxa as an indicator of biodiversity loss (Butchart et al., 2005). Changes in the index are driven by changes in category status of species on the IUCN Red List. As a consequence the RLI requires that species be assessed at least twice to be included. The RLI is gaining support as a useful biodiversity indicator at the national and global level and there is one example of its use at the provincial level (Quayle et al., 2007). However, the index has not been calculated for forest management areas and it is unlikely to be done for the following reasons. There are a relatively small number of species listed as threatened or endangered in Canada (389 as of June 1, 2009, Government of Canada, 2009) and this list is further reduced when dealing with provincial or regional forest management scales (for example, only 45 of the 389 species listed as threatened or endangered by COSEWIC are found in Alberta). Of the species within an FMA, there may be robust scientific information on population size and distribution for only 1–2 species. It seems highly unlikely that the change in status of so few species could be indicative of species diversity as a whole or as an indicator of ecosystem function. There is the additional problem that threatened and endangered species have specific management actions directed toward their recovery. This creates single species management remedies that further decouple the indicator from the broader "biodiversity" it is alleged to represent.

3.3. Trends in selected species distribution and abundance

Forest companies invest considerable time and money in counting species under the premise that they are monitoring distribution and/or abundance. Government agencies conduct similar monitoring in forested regions. However, despite attempts to diversify the number and type of species that are monitored there continues to be a strong bias toward vertebrates and those species that are exploited by humans (Shank, 1999; Clark and May, 2002; Dobson, 2005). For example, provincial agencies survey ungulates to obtain population estimates or relative abundance to set harvest quotas. These surveys tend to be highly variable from year to year depending on funds and perceived priorities. Special interest groups such as Ducks Unlimited conduct well-designed waterfowl surveys each year (e.g., North American Waterfowl Management Plan, 2009). Forest companies may collaborate with these agencies but they rarely conduct their own population assessments nor do they report on these species at an FMA level.

The taxon most frequently monitored by forest managers likely is song-birds. Selected species often are identified as "indicators" of some component of concern such as the amount of old growth, large standing dead trees, or un-salvaged burn habitat (Hobson and Schieck, 1999; Hannon and McCallum, 2003). The indicator species chosen vary between FMAs. In general, song-birds are monitored by point counts whereby birds heard and seen during a particular duration and distance are recorded by an observer (Ralph et al., 1995). Despite pleas for standardization of sampling duration and distance, considerable variation exists between and within FMAs and reflects in part regional variation in what the "optimal" method is for surveying birds (University of Alberta, 2009). Spatial and temporal sampling design is highly variable and tends to be linked to directed research programs covering relatively small spatial (up to a few km^2), and short time (1–3 years with a few studies spanning >10 years) scales usually in conjunction with stand-level experimental treatments. Such studies can give some indication of the effects of various harvest practices over short time frames but they are unlikely to track changes at the FMA-scale because sampling is usually biased by such things as ease of access and habitat quality (researchers tend to work where sample sizes can be maximized for the least effort, Anderson, 2001). Further, species that are relatively common are more likely to be studied for similar reasons. There are efforts to combine information collected by different researchers into broader regional databases (e.g., www.borealbirds.ca, University of Alberta, 2009) but there are major challenges getting access to data and standardizing variables. Forest birds are also monitored through volunteer-based programs like the Breeding Bird Surveys (e.g., Sauer et al., 2008) which apply consistent protocols over longer time frames and broader geographical areas. However, survey routes tend to be restricted by access, leaving large tracts of forest with no sampling effort (www.borealbirds.ca, University of Alberta, 2009).

There are other volunteer initiatives that collect occurrence records for a diverse range of species including owls (Bird Studies Canada, 2009), earthworms (Nature Watch, 2009), and butterflies (North American Butterfly Association, 2009). However, the scale, intensity and geographic range of sampling by these groups are highly variable depending on the energy and dedication of the individuals involved. In most cases, information is only recorded where species are observed with little to no record of spatial and temporal sampling effort. Few records exist for areas away from main roads and settled areas.

In recognition of the patchwork of agencies and initiatives collecting data across Canada, efforts are being made to pull information from all available data sources to report on species diversity at provincial and federal levels. Canada has produced two Wild Species reports that summarize the status and trend of species ranging from vertebrates to vascular plants (Canadian Endangered Species Conservation Council, 2001, 2006). A testament to the increased effort expended for the second report is the increase in coverage from 1670 species in 2000 to over 7700 species in 2005, as well as the increased breadth of taxonomic coverage. The Wild Species reports use status categories similar to COSEWIC, where species are placed into categories based on risk of extinction. Reports are produced every 5 years and changes in status of re-assessed species are meant to serve as an indicator of biodiversity status. Wild Species documented a number of shifts in status from 2000 to 2005 but the authors point out that 94% of these shifts were due to changes in information quality or methodology rather than to true, biologically based shifts in status (Canadian Endangered Species Conservation Council, 2006). This "data quality" problem also applies to COSEWIC assessments and it emphasizes the difficulty of drawing on such diverse information sources. Along a similar tact, the World Wildlife Fund has developed the Living Planet Index (LPI), which uses data on changes in population size over time of 1686 vertebrate species (both threatened and non-threatened) as a surrogate for biodiversity (Loh et al., 2005; World Wildlife Fund, 2008). To our knowledge, individual forest management companies have not attempted to report on changes in distribution and abundance of selected species using the procedures outlined by Wild Species or the LPI.

3.4. Summary

Current forest biodiversity monitoring in Canada is not sufficiently developed to provide timely, scientifically rigorous, consistent information to produce indicators that would allow us to effectively manage biodiversity. The major shortcomings can be summarized as follows:

1. Through the use of forest inventories, the status and trend of a limited number of habitat types designated by focal species and linked to tree species type, density, and age can be tracked. Habitats for most forest-dwelling species cannot be tracked and habitat definition is not standardized, making cross-FMA comparisons difficult.
2. Status and trend assessments of selected species remains heavily biased toward vertebrates and threatened species.
3. The fundamental information used for assessing the status of species (changes in distribution and abundance) is often so poor that changes in status are caused by changes in data quality rather than real changes in extinction risk.
4. There is no standardization of indicators, monitoring protocols, or sampling designs leading to any available information being FMA specific.

4. A new approach to forest biodiversity monitoring in Canada

4.1. A cumulative-effects monitoring approach

Given the shortcomings of current biodiversity monitoring what is the best way to improve the system? One option would be to simply devote more effort toward pulling together the current mosaic of biodiversity monitoring data. We suggest that this is unlikely to be effective by itself and advocate a major shift in approach beginning with the basic paradigm upon which the monitoring program is built. Virtually all forest biodiversity monitoring programs currently operate within a stress-oriented paradigm, where the goal is to detect the ecological effects of specific stresses (e.g. particular human activities) on particular indicators (Thornton et al., 1994; Trexler and Busch, 2003). Conceptual models often are used to describe the relationships between ecological stressors, processes, and indicators (Manley et al., 2000; Noon, 2003). This type of monitoring is well suited to small-scale, intimately understood systems because cause-and-effect relationships can be clearly identified (Thornton et al., 1994; Noon et al., 1999). Stress-oriented monitoring goes hand in hand with research programs designed to assess the effects of stand-level practices on biodiversity but this approach has major problems when multiple management effects such as cut levels, patch size, fragmentation, road density, fire suppression, and human access interact to affect biodiversity. Although it may be desirable and necessary to separate individual stressors for experimental purposes, it is the cumulative effects of all human activities that ultimately determine biodiversity change. Relying on controlled stress-oriented experimental studies to track FMA-level biodiversity runs the real risk of "missing the forest for the trees". As well, because this paradigm focuses on known relationships, it is less effective when novel, unanticipated stressors are introduced into the system (Maes and Van Dyck, 2005).

Cumulative-effects monitoring (termed surveillance monitoring by Nichols and Williams, 2006), on the other hand, is targeted at detecting the ecological effects of a diverse set of environmental stresses on broad suites of indicators. Rather than focusing on specific cause-and-effect relationships, cumulative-effects monitoring exposes correlative relationships between multiple stressors in a system and the many indicators that are monitored (Thornton et al., 1994; Noon et al., 1999). As such, cumulative-effects monitoring is more amenable to assessing progress towards broad management objectives like "maintaining biodiversity" on an FMA (Mulder et al., 1999). In addition, compared to the more narrowly focused stress-oriented monitoring, cumulative-effects monitoring is more likely to meet the needs of multiple organizations. Finally, cumulative-effects monitoring remains relevant over long time frames as new human activities and environmental stresses arise. Given that forest managers attempt to manage for sustained wood supply and biodiversity over long time frames and large spatial scales, that they use a diverse suite of practices to enhance biodiversity (stand-level practices to retain structure; landscape-level practices to vary the size and shape of cutblocks; protected areas to retain old growth), and that other land uses can and do occur in conjunction with forest harvesting, we think a cumulative-effects monitoring paradigm is most appropriate. This does not mean that there is not a role for stress-oriented monitoring, which provides important information about specific management practices. We wish to stress however, that project-based stress-oriented monitoring cannot be "added up" to capture biodiversity change at the scale of FMAs, provinces, or the country. We need to widen our focus from solely stress-oriented approaches to include the development of additional cumulative-effects-oriented biodiversity monitoring programs (see Alberta Biodiversity Monitoring Institute, www.abmi.ca) to strike a balance. Together, stress-oriented research and cumulative-effects-oriented monitoring provide a powerful tool set that can be combined with modelling to better predict human impacts on biodiversity at the scale of multiple FMAs.

4.2. At what spatial scale?

Although forest harvesting and silvicultural practices are prescribed at the stand level, the time and space scales of natural disturbances and forest succession require that forest management planning for wood supply and biodiversity be done at the scale of 100's to 1000's of km^2 over multiple decades. Biodiversity monitoring should be designed to deliver statistically powerful results at this scale also. This means we have to re-focus current programs which are usually designed for stress-oriented project monitoring at stand scales or are amalgams of these reported at national and global scales. It is undeniably useful if a core set of monitoring variables can be standardized across FMAs to allow scaling up of results and comparisons regionally and nationally (e.g., Pereira and Cooper, 2006). In fact, it will be crucial to do so as the added complications of climate change effects on biodiversity become more prominent. If biodiversity can only be tracked within an FMA (because indicators and protocols are not consistent), there is no statistical means to separate changes in biodiversity caused by the FMA holder's management strategy from changes caused by very large-scale phenomena like climate change. If biodiversity information was standardized and collected consistently both on and off FMAs (protected areas, non-productive forest for example) it would then be possible to compare between FMAs to assess the relative success of different management strategies (Adaptive Management Approach, Walters, 1986) and between FMAs and areas with low human use to determine some of the over-riding effects of climate change. Without such information, forest managers face the real risk of being blamed by the public for all of the negative changes in biodiversity that may occur on their FMA in the near future.

4.3. What should be measured?

We are in full agreement with the literature that coarse-filter surrogates such as extent and pattern of selected ecosystems, as well as pressure indicators such as extent of human disturbances, are invaluable components of a biodiversity monitoring program (Hagan and Whitman, 2006; Mace and Baillie, 2007). Where we differ is with regards to the species monitoring component. We will never be certain that the species we select to monitor are serving as an adequate thermometer for biodiversity as a whole but it is also highly likely that the current emphasis on vertebrates (representing <3% of all known species, Clark and May, 2002) and threatened species is inadequate. Given that forest management and forest landscapes change over long time frames and large spatial scales, our incomplete understanding of natural systems, continued extinctions and extirpations, and our inability to directly measure many ecosystems services requires that we include multiple taxa with varying life history traits and ecological roles. We simply cannot continue to ignore diverse taxa such as arthropods and species that perform important ecological functions (e.g., decomposers). This community "coarse-filter" (Parrish et al., 2003) or "bet-hedging" (Manley et al., 2004) strategy is a middle ground between the impossible (measuring all aspects of biodiversity) and the ineffective (using a limited indicator species approach). A limited number of indicator species is problematic because their surrogacy and sensitivity rarely are tested; when they are, indicator utility in even adjacent regions can be contradictory (Sergio et al., 2006; Roth and Weber, 2008). In addition, a small number of indicators tends to lead to single-species solutions that keeps conservation anchored firmly in a reactive, crisis conservation mode: we learn how to manage the stressor for that species, but do not necessarily learn how ecosystems change with human intervention or cumulative impacts.

We disagree with the perspective that only species that are known to be sensitive to existing human-caused impacts should be monitored (e.g., Dale and Beyeler, 2001): this concept arises from a perception of biodiversity monitoring as solely a "canary in a coal-mine" situation, where cause and effect are well-understood. Such an approach also assumes that we can predict future human-caused effects and history shows the contrary. We, like Gaston and Fuller (2007), disagree with the hypothesis that rare species are better biodiversity indicators than are common species. If we really want to monitor trends in biodiversity in response to forest management, then we should choose taxa from across the relative abundance spectrum, that occupy different trophic levels, and that have a range of body sizes and life histories. It seems highly unlikely that the current limited list of species indicators being monitored in our forests is serving as an effective thermometer of biodiversity as a whole.

4.4. What sampling design and protocols should be used?

The primary purpose of biodiversity monitoring programs is to provide environmental information that will help stakeholders evaluate the sustainability of human activities (Balmford et al., 2005). This requires that biodiversity monitoring programs collect unbiased information that has sufficient statistical power to detect changes important to stakeholders. Though it is often logistically inefficient, a random sampling design is the surest method to ensure that unbiased data are collected (Krebs, 1989). As long as there is no periodicity in the indicators being monitored, systematic sampling also produces unbiased data and has the added benefit of ensuring that the samples are spaced throughout the region (Krebs, 1989). More complex stratified, neighbourhood or reduced-effort sampling designs may achieve higher statistical power with the same sampling effort as compared to random or systematic designs (e.g., Thompson, 2004; Roy et al., 2007), but a number of constraints limit their use for large-scale, long-term biodiversity monitoring. Different stakeholders often have different regions of interest (e.g., a specific ecological region, a particular industrial management region, or a political administrative region) that may only partially overlap each other. Complex sampling designs can be optimized for the total region, or for any portion of the region, but cannot be simultaneously optimized for the region as a whole and all portions of the region. More importantly, statistical power of complex statistical designs decreases over time as political, industrial, and ecological boundaries change (Edwards, 1998). Thus, although statistical power is not maximized at the start of a monitoring program in random or systematic designs, these designs support multiple stakeholder needs and power does not deteriorate over time.

Statistical power of a monitoring program is positively related to the number of locations being sampled and the frequency with which each location is re-sampled (Field et al., 2004). However, spatial and temporal variance in the data being collected also affects statistical power. It is possible for biodiversity monitoring programs to statistically control for a portion of the spatial variance by surveying the same locations repeatedly over time (Urquhart et al., 1998; Larsen et al., 2001). In addition, for a given sampling effort, creating a series of panels (geographic divisions; e.g., five panels with 20% of the locations in each panel) and surveying one panel each year, achieves higher statistical power than having fewer locations and surveying all locations each year (Urquhart et al., 1998; Urquhart and Kincaid, 1999). However, if resource managers know the locations that will be surveyed, they may intentionally, or unintentionally, choose to modify activities at those locations. Biased activities erode the value of the monitoring program because the data become less representative of the region. Thus, if monitoring programs resurvey the same locations over time, we recommend that the locations of monitoring points are not revealed to forest managers.

Statistical power is also related to the magnitude of change that stakeholders wish to detect. By including indicators that respond dramatically to environmental change or combining information among multiple indicators and creating composite metrics that provide very strong signals of ecological change it is possible to increase statistical power (Karr, 1981; Foster, 2001; Maes and Van Dyck, 2005). Minimizing measurement error also is key (Seavy and Reynolds, 2007), and can be achieved by including surrogates with a high signal-to-noise ratio, surveying those surrogates during a consistent seasonal period, and by training technicians to conduct methods consistently.

Optimizing statistical aspects of a monitoring program also requires feedback from stakeholders about their needs and the costs they are willing to incur (Dobson et al., 1999; Green et al., 2005; Maxwell and Jennings, 2005). Costs of a monitoring program are directly proportional to sample intensity (number of sites sampled and resample frequency), while statistical power increases asymptotically with sample intensity (Nielsen et al., 2009). At the optimal sample intensity, stakeholders will conclude that the marginal increase in statistical power is not worth the increased expense (Millard and Lettenmaier, 1986). As part of the optimization process, α, β, and effect size can be varied to assess how changes in these affect power and costs (Carlson and Schmiegelow, 2002). In addition, the optimization process can be used to highlight the statistical power that will be achieved for each surrogate, and help stakeholders decide which should, or should not, be included in the program.

4.5. Who should do the monitoring?

It may not be important to specify whether government agencies, industry, or arms-length organizations do the monitoring. If data collection and sampling protocols were standardized, information could be collected by any of these groups. The important point is that it has to be someone's business to produce the sampling design, the protocols, and the way in which information is brought together and summarized. In addition, data should be freely available to ensure scientific legitimacy and transparency. Biodiversity monitoring and reporting is extremely complicated as evidenced by the effort and time it has taken organizations like the World Wildlife Fund and the Convention on Biological Diversity to develop the analyses and reporting protocols they currently use.

4.6. Who should pay?

Large-scale biodiversity monitoring programs are expensive. They need to collect, store, analyze, and communicate large amounts of data (Green et al., 2005). All the while, they need to ensure that the programs remain scientifically credible, and produce information that is valuable to stakeholders (see discussion above). Further, programs may need to operate for a decade or more before sufficient statistical power to satisfy the needs of stakeholders is achieved (Watson and Novelly, 2004). This means that monitoring programs require considerable investment during their early years when their products are less attractive to stakeholders. Given the short-term budgetary cycles of governments, industries, and environmental organizations, we recognize that securing long-term funding is difficult—it is perhaps the toughest challenge for biodiversity monitoring as we describe it. Sufficient long-term funding, however, ensures that the program can attract and hold quality staff, collect high quality information, and develop effective communication tools (Caughlan and Oakley, 2001). Biodiversity monitoring programs that cannot secure funding that is robust to changing political, economic, and corporate conditions probably should not be initiated (Watson and Novelly, 2004).

That being said, there are currently large sums of money being devoted towards the broad objective of biodiversity monitoring by industry and provincial governments. Given the poor performance of this investment to date, it seems appropriate that many of these funds could be re-assigned to a broad-based collaborative, consistent biodiversity monitoring program where data could be shared by all interested parties. A comprehensive assessment of current expenditures in Alberta suggested that as much as 80% of program costs could come from re-directed funds, much of which otherwise would be spent on project-specific, small-scale, 1–2 years monitoring projects (Alberta Biodiversity Monitoring Program, 2006).

5. Conclusions

We think that the simple question of "How is biodiversity changing in forests managed for long-term wood production?" is an important one that strikes at the core of the claim that these regions are managed with both biodiversity and timber values in mind. Although forest companies and government agencies clearly have the means and information to report on the amount of forest grown and harvested, they do not have a similar ability to report on biodiversity change. The current ad hoc reliance on a patchwork of inconsistent protocols, sampling designs, and research-oriented studies means that the question we posed at the outset cannot be answered. We recognize that some of our recommendations are controversial, and that large-scale, relatively expensive programs (such as the Alberta Biodiversity Monitoring Institute, www.abmi.ca) will always be heavily scrutinized. However, given the nature of biodiversity and the diversity of human impacts, if forest management agencies are sincere in their desire to manage biodiversity, they need to devote the same effort and scientific rigour used to monitor tree growth and harvest rates to developing standardized protocols and rigourous sampling designs for biodiversity monitoring.

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