

Modelling and monitoring: examining the utility of dynamic landscape metapopulation models for sustainable forest management

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Abstract

Reliable, practical and affordable means of assessing the sustainability of forest management remain elusive. Monitoring of biological indicators is an important element, but sufficiently powerful monitoring strategies are expensive and monitoring alone may not provide answers in time to avoid irreversible environmental or ecological damage. We propose a model-based approach to assessing sustainability using bio-indicators of ecosystem condition to provide timely feed-back to managers about the sustainability of current and alternative forest management options, but acknowledge that assessment of biological indicators is only one component of the assessment of sustainability. This model-based approach can also support the development of better-targeted and more relevant monitoring systems.

Dynamic landscape meta-population (DLMP) models integrate spatial models of forest change (also known as landscape dynamic models or forest succession models) with meta-population models, which describe demographic and biological attributes of species and the dynamic consequences of migration and habitat change. DLMP models may be used to predict the meta-population level consequences of management actions on bio-indicator species, and as such, may provide a relatively efficient and inexpensive approach to assessing an important aspect of the potential sustainability of forest management.

We review some of the criticisms of monitoring for trend in indicators and the advantages and problems associated with using DLMP models of bio-indicator species to evaluate the sustainability of forest management options. We draw on results of a

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case study of the brown creeper (*Certhis americana*) in northern Ontario, Canada, that explored the sustainability of competing forest management scenarios. Based on case study results, DLMP models of bio-indicator species appear to be useful for (1) assessing and ranking the sustainability of management options, (2) quantifying the stresses placed on ecosystems by particular management activities, (3) targeting future research and data collection, and (4) dealing explicitly with both environmental and model uncertainties.

Introduction

Approaches to assessing the sustainability of forest management often include the long-term monitoring of indicators for evidence of change and implied impact (Canadian Council of Forest Ministers 1995). Elements of forest biodiversity are usually included in a suite of indicators to be monitored including small mammals (McLaren et al. 1998), birds (Venier and Pearce 2004) and salamanders (Welsh and Droege 2001) that have all been examined for their potential to address this issue. Within an adaptive management framework, the results of monitoring should be used to modify management practices in an iterative way that should lead to more sustainable practices (Walters and Holling 1990). In this paper we discuss why this paradigm may not be adequate to address the unique issues of sustainable forest management and we describe an alternative modeling based approach that can address some of these criticisms. We use results from a case study of a forest bird (brown creeper, *Certhia americana*) in an actively managed forest to demonstrate the modeling approach and its relationship to monitoring.

Monitoring for Trend

In the context of forest management planning, there are at least 3 reasons why long-term monitoring cannot adequately connect bio-indicators to planning for forest sustainability. The first is that forest management planning is complex and time consuming, while forest change due to harvesting can be relatively rapid. Forest management plans require years to develop and implement, and much time is also required to develop policy from the results of monitoring and research. Large investments in time and money are also required to achieve reasonable statistical power to detect trends in bio-indicators. For example, Schalk *et al.* (2002), based on data collected using standard point count protocols (Welsh 1995), estimated that for bird species occurring at 50 sites, 10 years of annual sampling would be required to provide 80% power to detect an 18-26% decline. These time frames are too slow to provide useful feedback on impacts of forest management plans.

Second, it is difficult to establish that real and important changes are occurring in populations using broad-scale monitoring programs (Temple and Wiens 1989). Broad-scale programs are suited for detecting changes over long time periods and over large areas. Forest management is conducted at multiple spatial scales, but principally it is conducted at the scale of stands for operational prescription and at the scale of management units for broader objectives. Thus, evaluations of success or failure must occur at these relevant spatial scales.

Third, population trends, in and of themselves, do not help identify cause-and-effect relationships and therefore do not help identify management options (Ralph *et al.* 1995). For example, Morrison (1986) suggested that birds respond to change, but not uniquely to specific changes, therefore the cause of change is difficult to identify unless the data are collected within carefully designed experiments. This is true for all of the taxa that might be considered as indicators. The general lack of controls in large-scale monitoring programs is problematic because of the pervasive impacts of large-scale disturbances, such as climate change. Impacts that are not the result of forest management need to be factored out. Information about underlying causes of change are more likely to be obtained through intensive local-scale studies.

Habitat Supply Monitoring

Recently, as an alternative to monitoring for trend, habitat supply modeling has been used to examine the impact of forest management plans on selected wildlife species (Voigt *et al.* 2000; Mulder *et al.* 1999). In this approach, the habitat of an indicator species is assessed through small scale focused studies and expert opinion, to arrive at a map of habitat quality across the landscape. These maps are used to measure future habitat supply under alternative forest management scenarios. However, habitat suitability models do not consider environmental and demographic stochasticity, or metapopulation dynamics that can have profound impacts on the viability of populations (Burgman *et al.* 1993).

Dynamic Landscape Metapopulation Modelling

A more holistic approach to examining management impacts on indicator species would be to link dynamic landscape models describing habitat availability through time with spatially explicit models of population viability over time. Dynamic landscape metapopulation (DLMP) models integrate spatial models of forest change with metapopulation models, which describe demographic and biological attributes of species and the dynamic consequences of migration and habitat change. DLMP models may be used to predict the meta-population level consequences of management actions on bio-indicator species, and as such, may provide a relatively efficient and inexpensive approach to assessing an important aspect of the potential sustainability of forest management. DLMP models could become an important link in the adaptive management process. Sensitivity analysis within the modelling process could help to target validation monitoring for the refinement of model parameters. Effectiveness monitoring would be used to check model predictions in the context of carefully designed experiments. We implemented a DLMP case study using RAMAS Landscape (Akçakaya *et al.* 2003) which is comprised of two separate modules: RAMAS GIS (Akçakaya and Root 2002), which simulates a species' metapopulation dynamics over time, and LANDIS (Mladenoff and He 1999), which simulates forest change by modeling tree species composition in ten-year age classes.

Four Steps in Model Development

To examine the utility of DLMP models as a means of incorporating bio-indicators into forest management planning, we conducted a case study with the brown creeper (*Certhia americana*) in an actively managed study area in the boreal forest of Ontario (Wintle et al. in press). We chose the brown creeper because it depends on large trees for foraging and nesting and is associated with a closed canopy and snags (Hejl et al. 2002). Theory suggests that for efficiency, indicators should be chosen to target known stresses on the system (Mulder et al. 1999) and timber harvesting truncates older tree cohorts and reduces the number of snags (McRae et al. 2001).

There is insufficient scope in this paper to provide the details of the modelling exercise but we will briefly review the steps required in model development and provide some examples of outputs and interpretations for the case study. Details can be found in Wintle et al. (2005). The general implementation of DLMP models requires 4 steps outlined below.

Step 1: Building a habitat model

Observational data, expert knowledge and literature review is used to develop a mathematical model describing the relationship between occupancy by the indicator species and environmental variables such as forest composition, topography and climate. In the case study, we explored these relationships using logistic regression analysis. Environmental variables are restricted to those for which mapped data exist so that the model can be used to predict the locations of suitable habitat across the whole study area. For example, in the creeper case study, regression analyses strongly supported the inclusion of a positive relationship between creeper occupancy and stand age ($p < 0.05$), explaining approximately 15% of the variation in observation data. Expert opinion and prior studies supported the exclusion of pure stands of jack pine and black spruce from suitable habitat. Occupancy by the creeper was then predicted across the entire study region using raster maps of forest tree species composition and age that were based on forest resource inventory maps.

Step 2: Developing a population model

A stage-structured population model (Akçakaya and Root 2002) is constructed using RAMAS GIS. The stage matrix is based on available information about survival, fecundity and the probabilities of transition from each stage to the next. In many cases, accurate estimates are not available for some of the required vital rates and their standard deviations. In this case, best estimates are made and their influence on model predictions can be tested by sensitivity analysis. Other parameters that must be considered are the form of density dependence and competition.

Step 3: Linking these models in a metapopulation model

To link the habitat model to the population model, the spatial structure of the populations must be specified, and the carrying capacity, dispersal, and spatial autocorrelation in demographic stochasticity must be quantified. RAMAS GIS identifies discrete patches of connected habitat from the habitat suitability map developed in step 1. Patches are treated as discrete populations. The carrying capacity (K) and initial number of individuals in each patch is estimated as a function of patch habitat quality. During model simulations, K is sampled from a distribution with mean equal to the estimated K and standard deviation set by the user. The initial abundance of each patch is set to K , unless there are grounds for assuming that the species was recovering from a period of

low habitat occupancy. The probability of dispersal between the edges of any two patches is a function of the distance between patches. The dispersal function plays an important role in metapopulation models because it determines how quickly patches can be colonized by immigrants from other populations (Akçakaya 2000). Spatially structured populations tend to experience spatial autocorrelation in fluctuations of vital rates because the environmental influences on vital rates are themselves spatially structured (Akçakaya and Root 2002). Often, little information is available for specifying the autocorrelation function, though the sensitivity of the model to alternative assumptions about the form of autocorrelation should be tested.

Step 4: Building a forest dynamics model and linking it to the metapopulation model

The LANDIS model (Mladenoff and He 1999) was used to generate predictions about the future structural and floristic composition of the forest under a variety of management scenarios. In the creeper case study, four silvicultural scenarios were simulated in LANDIS based on various combinations of mean harvest size, tree retention rates and salvage logging practices (Table 1). Each scenario was subject to the two alternative fire regimes. A separate LANDIS model was developed for each of the eight combinations of harvest and fire scenario and was used to predict the future floristic composition and age of the forest over 160 years. These predictions were used as a basis for the creeper DLMP model, which was run for 160 years, using 1000 replicates.

To generate these scenarios, succession dynamics of the dominant tree species are modeled. Estimates of life history parameters including longevity, seeding distance, age at sexual maturity, shade tolerance and fire tolerance must be provided. Initial species composition and age class distribution must be specified. In the case study this starting condition was derived from historical inventory data. To simulate fire, mean fire return times and a fire size distribution are estimated. The LANDIS module generates predictions of forest species composition and structure as a time series of raster maps (1 ha grid cell size) at 10 yearly intervals for the duration of the simulation (160 years in this case, representing the length of two harvesting rotations). Outputs include a time series of raster maps representing forest species composition and the age of the dominant strata in each 1ha cell. RAMAS Landscape links the metapopulation model and the forest dynamics model by creating a time series of habitat maps based on forest age and composition maps produced by LANDIS.

Modelling Outcomes

Habitat Availability

The impact of each scenario (Table 1) on the amount of available habitat can be expressed in terms of changes in carrying capacity (K) in the study region resulting from each management scenario. K is linearly related to the amount and quality of habitat in the region. Figure 1 shows that, in the creeper study, K declined substantially under the most intensively managed scenarios that involved salvage logging and conversion of logged areas to jack pine plantation (scenarios 7 & 8). The variability in available habitat was more strongly a function of alternative fire regimes than alternative harvesting.

Table 1. Forest Management Scenarios: Forest composition and structure at the first time step reflects that of 1970 according to historical forest inventory maps.

1. No timber harvesting and a natural fire regime

Fire return times: 110 years for jack pine forest, 350 years for mixed forests, mean fire size 2000ha, an upper and lower limits of 8000 and 200ha.

2. No timber harvesting with fire suppression

Fire return times: 320 years for jack pine forest, 700 years for mixed forests, mean fire size 8000ha, an upper and lower limits of 10000 and 6000ha.

3. Natural disturbance emulation harvesting guidelines with natural fire regime

20% of the harvested area in the region is allowed to regenerate naturally to mixed forest, with the remainder replanted to jack pine. 10% of the stands nominated for harvesting are retained in one hectare blocks as wildlife habitat, and are not harvested. The total area harvested is approximately 18000ha in each of two rotations.

4. Natural disturbance emulation harvesting guidelines with fire suppression

5. Intensive harvesting with a natural fire regime

All areas nominated for harvesting are clearcut and replanted to jack pine. All replanted areas remains as jack pine for the duration of the simulation.

6. Intensive harvesting with fire suppression

7. Intensive harvesting with natural fire regime and salvage logging in burnt loggable areas

Areas burned by wild fire are then salvage logged. 20% of the salvage-logged area is allowed to regenerate naturally and the remainder is planted with jack pine. No salvage logging occurs in Pukaskwa National Park.

8. Intensive harvesting with fire suppression and salvage logging in burnt forest under an altered fire regime

Population Trajectories

Population trajectories can be constructed and represent population size fluctuations over the simulation period (see Figure 2 for an example from the case study). Examination of the mean tendency for the base scenario (no timber harvesting and no fire suppression; top left box in Figure 2) reveals that the brown creeper population in the study area is predicted to be relatively stable over 160 years, fluctuating around a mean

population size of approximately 80,000 individuals. Trajectories for the brown creeper under alternative management scenarios differed from the base model, with declines predicted as the intensity of disturbance increased. For example, the expected minimum population (EMP) of scenario 8 was 32% lower than the base scenario. Under the assumptions made in the model, the current style of forest management (most closely resembled by scenarios 4 and 6) resulted in an approximately 24% decrease in EMP compared to the base scenario. However, only 11% – 14% of the landscape was harvested in scenarios 4 and 6, which suggests that metapopulation dynamics are influencing the result, due to the impact that harvesting may have on dispersal rates. Under most scenarios, the predicted minimum population size was not in direct proportion to the change in carrying capacity over the simulation, indicating that population processes, beyond simple habitat availability, were influencing model results.

Sensitivity Analysis

Sensitivity analysis is conducted to examine the sensitivity of the model outcomes to different scenarios, estimates of model parameters, their standard deviations and other model assumptions. We examined sensitivity by calculating the change in expected minimum population (EMP) size (McCarthy and Thompson 2001) that results from a change to a single parameter. In our case study, the difference between the base model EMP and alternative scenario EMPs tended to increase with increasing intensity of disturbance. All alternative scenarios show negative sensitivities when compared with the base model, with the highest impact scenario having by far the greatest negative impact on EMP with a sensitivity of -31.64%.

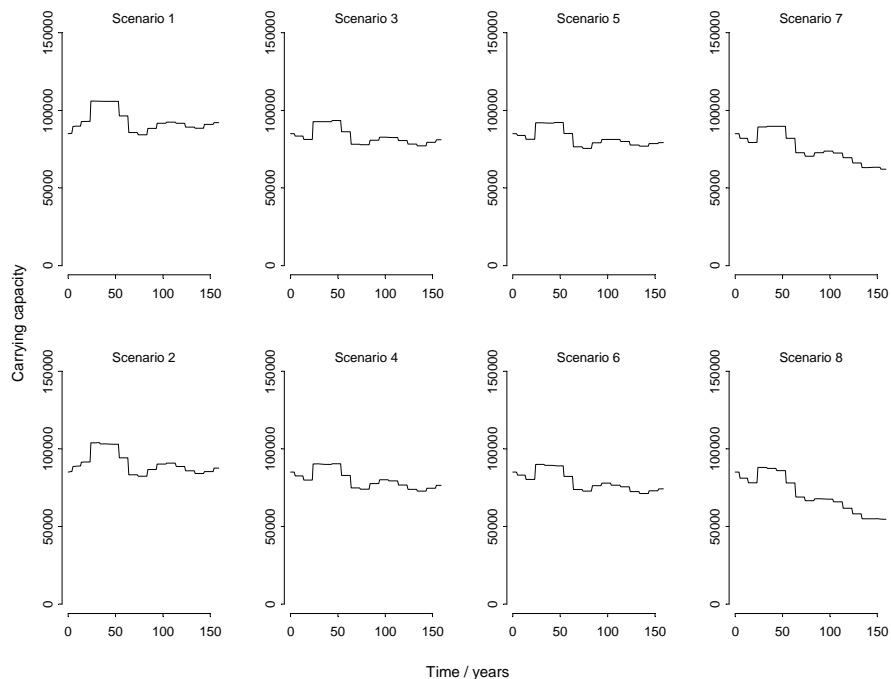


Figure 1: Changes in carrying capacity over 160 years under each management scenario. Carrying capacity (Y-axis) is the total carrying capacity over all populations in the region.

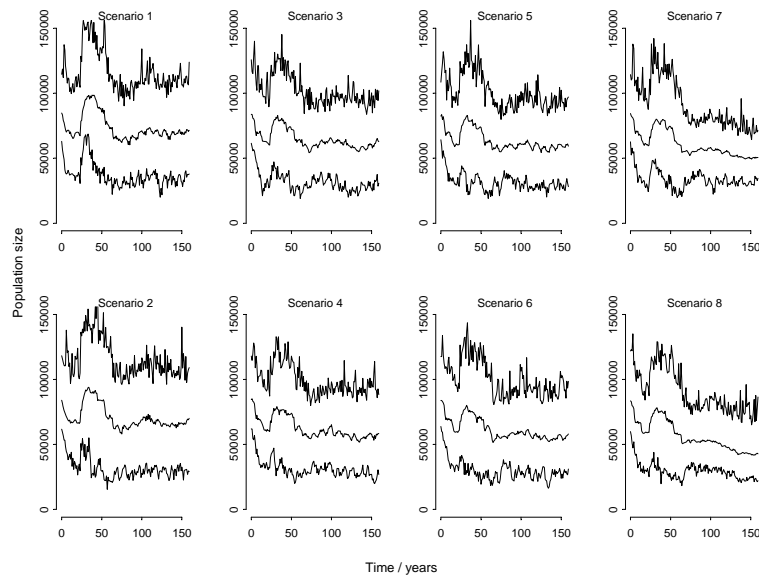


Figure 2: Trajectory summaries for the brown creeper population model under eight alternative forest management scenarios. On each plot the middle line represents the mean population size of 1000 replications. The upper and lower lines represent the maximum and minimum population size in 150 simulations (3 population simulations on each of 50 landscapes).

Discussion

We suggest DLMP models as an approach for incorporating biological indicators into the sustainable forest management planning process. Forest management is an extremely complex activity at a local and landscape scale and is confounded by environmental changes at regional and global scales. It can also result in very rapid change to the land base especially relative to the rate at which we can assess changes in biodiversity. A proactive, modeling approach addresses many of these issues.

We provided an example of DLMP model development using the brown creeper in an actively managed boreal forest area. The model predictions for brown creeper habitat and populations were sensitive to the differences between management scenarios, indicating that these methods are useful for assessing and ranking the sustainability of forest management options. As well, the model results suggest that the demographic and spatial components of the model do influence the outcomes. DLMP model results provide transparent statements of the predicted relative cost of management actions in terms of population outcomes. This is information necessary for rigorous weighing of costs and benefits of alternative management options.

The DLMP approach has advantages over long-term monitoring programs for example, the relatively low cost, the immediacy of the information, the explicit treatment of uncertainty and the flexibility of the method to provide predictions about future management scenarios. Models may provide an early warning of unsustainable practice where monitoring cannot. However, model based approaches must be supported by monitoring studies because the latter are critical to developing biological knowledge,

testing predictions, and refining parameters (Walters and Holling 1990). Validation monitoring is data collection designed to test assumptions of the model or document cause-and-effect relations (Noss and Cooperider 1994). Almost all models in resource management are built with insufficient information about parameters necessary to the model. As discussed, sensitivity analysis can be used to identify those assumptions that matter most to the final model outcomes. Therefore, sensitivity analysis is an efficient and transparent method to ensure monitoring is targeted to address the most important questions first. Effectiveness monitoring is data collection to measure the effectiveness of the management approaches being taken (Noss and Cooperider 1994). Models predict the impacts of certain management options but these predictions need to be checked with empirical data. In the best circumstances these predictions would be examined in the context of an experiment designed to highlight impacts.

The applicability of these methods will vary from species to species depending on life history and habitat specificity. DLMP modeling has generality in that it has the potential to be applied to many species that utilise a range of habitats at different spatial scales. An important advantage of model based approaches is that they allow numerous species to be modeled to provide a representative sample of the ecosystem response to management. Future implementation of model based approaches will require a means to aggregate results of several species models to enable a multi-species evaluation of management options or the state of the system.

RAMAS GIS appears well equipped to model the various forms of uncertainty associated with demographic parameters. The creeper model was not particularly sensitive to 20% changes in most population parameter estimates except an alternative assumption about the mechanism of density dependence. However, the relative ranking of the scenarios did not change in response to alternative assumptions for these parameters (B. Wintle, unpublished data), implying that the general result was robust to population model assumptions. RAMAS Landscape, however, does not provide a simple means for incorporating the uncertainty in model predictions associated with landscape model assumptions and parameter estimates such as the fire size distribution and fire frequency. This is clearly a future research priority.

Conclusions

DLMP models of bio-indicators may provide an efficient approach to assessing the potential sustainability of forest management by being inexpensive to implement, providing immediate information, explicitly dealing with uncertainty and providing predictions about future management scenarios. The DLMP model focuses monitoring efforts by identifying important knowledge gaps; highlighting both environmental and model uncertainties, and incorporating them directly into the decision making process; making the causal linkages between management actions and indicator response clear; and allowing forest managers to compare the sustainability of management options. The models must be one component of an adaptive management system in which the results of monitoring are used iteratively to refine model parameters and predictions.

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