Wildlife Habitat Management

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Abstract Wildlife biologists have been designing habitat models for over 50 years; however, the use of the Ecosystem Management Decision Support system (EMDS) in association with modeling is a relatively recent addition to the field. EMDS has proven its usefulness to habitat modeling and evaluation through successful application to a number of large landscape (>5000 km^2) studies. While EMDS cannot be used to model wildlife populations directly, past efforts have incorporated population data, along with a variety of other indicators. Here, we provide an overview of previous modeling efforts, with an emphasis on the northern spotted owl (*Strix occidentalis caurina*). We then review applications of EMDS to wildlife modeling, before delving into a case study. In that study, the Washington State Department of Natural Resources (DNR) used EMDS to assess the impacts of alternative forest management strategies on dispersal habitat for spotted owls. We discuss how expert workshops were used to define three separate EMDS models to assess and score patches of foraging, roosting, and movement habitat across a large landscape. We used the habitat scores to develop a dispersal habitat model outside of EMDS, which incorporated graph theory concepts and a variable resistance landscape surface to assess the connectivity of owl dispersal habitat.

Keywords Wildlife habitat · Habitat management · Northern spotted owl · Dispersal habitat · Spatial decision support · EMDS

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1 Historical Background

Responsibility for wildlife management in the United States is shared between states and the federal government. The federal Fish and Wildlife Service (USFWS) has the regulatory responsibility for protecting terrestrial and aquatic species, as mandated in the Endangered Species Act (ESA 1973) and other laws. State agencies work within federal guidelines (and supplementary state regulations) to do most of the day-to-day wildlife management. It is within this context that, in 1997, the Washington Department of Natural Resources (DNR) signed a habitat conservation plan (HCP) with the USFWS. The HCP is a relatively recent legal mechanism, requiring land managers to commit to a conservation plan in exchange for certainty that no further requirements will be imposed by the USFWS for the duration of the agreement (70 years in the case of the DNR). The DNR HCP addresses two federally protected terrestrial species, the northern spotted owl (NSO, *Strix occidentalis caurina*) and the marbled murrelet (*Brachyramphus marmoratus*), salmonid habitats and those of a number of other late-successional and old growth associated species of concern. It covers approximately 650,000 hectares of state forest lands in western Washington within the range of the NSO. The DNR's conservation objective is to provide habitat that makes a significant contribution to demographic support, maintenance of species distribution, and facilitates dispersal for NSOs.

By state law, the DNR is required to set a sustainable harvest level for state-owned forest trust lands, which they completed in 2004 using a coarse statewide analysis (WADNR 2004). The state's trust responsibilities support public education in Washington. They are also required to develop detailed tactical forest plans for each HCP planning unit. The DNR began this process with their South Puget Planning Unit, where they manage approximately 145,000 acres within a broader context of private and federal lands (Fig. 1). As specified in the HCP, management on DNR lands should be coordinated with management of federal species conservation areas. Areas within two miles of federal late-successional reserves (LSRs) were reviewed for designation as NSO nesting, roosting, and foraging habitat (NRF), while lands falling between large federal reserves were given special consideration for dispersal habitat. Two areas within the HCP South Puget Planning Unit, the Elbe-Tahoma and Black Diamond, contain approximately 80 % (66,000 acres) of the designated dispersal lands defined in the HCP.

Management efforts preceding the development of the South Puget Forest Land Plan Environmental Impact Statement had revealed a number of limitations to the original HCP dispersal habitat definition. First, it used a simple threshold approach, so that any stand evaluated either meets or does not meet the standard. This approach did not recognize stands meeting several but not all criteria. Second, initial dispersal definitions were considered minimal criteria for meeting the needs of dispersing owls, and more recent thinking questions whether they provide adequate structural complexity, especially to support foraging during dispersal (Buchanan 2004). Finally, the definition was weak on landscape configuration. It required that

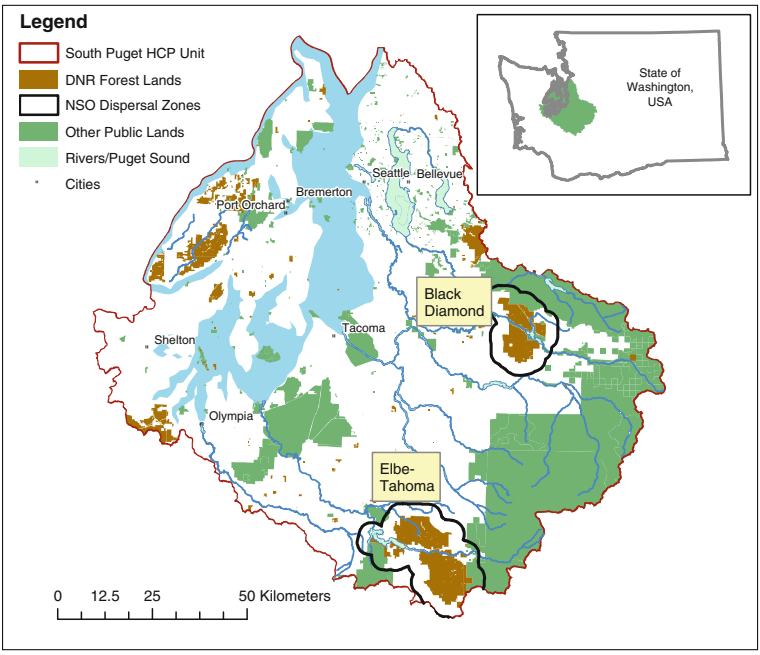


Fig. 1 Northern spotted owl dispersal areas analyzed within the South Puget HCP planning unit

50 % of each watershed unit achieve dispersal standards, but it was not clear how well such a requirement would actually support dispersal. These weaknesses, as well as a mandate for investigating ways to improve owl management in the HCP, caused the DNR to search for new methods to assess habitat for the NSO, especially dispersal habitat. Above all, the DNR needed a standardized tool that could be used to compare results of habitat assessments under multiple management scenarios over time.

2 Modeling Applications in This Problem Domain

2.1 Wildlife Modeling

Wildlife management has been defined as the "manipulation or protection of a population to achieve a goal" (Caughley and Sinclair 1994 p. 1). Predicting how populations might respond to different management options is often complex, because it is dependent on a variety of demographic and habitat relationships. This complexity has led to the development of a wide variety of computational models. In this section, we review some of the basic elements of wildlife modeling and

summarize efforts related to the NSO, to place our case study within this broader context and discuss the utility of EMDS in this domain.

Models for wildlife management, as in other disciplines, can fulfill a number of purposes, including defining and formalizing problems, clarifying ideas, organizing concepts, identifying knowledge gaps, communicating information, developing and testing hypotheses, and making predictions (Morrison et al. 2006). Researchers and managers have developed a wide variety of modeling approaches and structures to meet this range of objectives. Roloff et al. (2001) broadly classified wildlife models into three categories: habitat models, population models, and spatial population models.

A further distinction in habitat modeling is the implementation of coarse versus fine filter approaches. Coarse filter approaches look at the distribution of ecological community types over a landscape, often in relation to some historical baseline, while fine filter approaches characterize habitat availability for specific species (Noss 1987). Conversely, individual species are sometimes used as indicators of broader ecological communities or habitat types, as has been the case with the NSO, often thought of as an indicator species for the condition of old-growth forests in the Pacific Northwest (Dawson et al. 1987; Lee 1985; WADNR 2010b). The indicator species concept has been recently criticized, and the validity of various refinements is hotly debated (e.g., see Cushman et al. 2010; Lindenmayer et al. 2002; MacNally and Fleishman 2004; Simberloff 1998; Wiens et al. 2008).

Probably the earliest type of wildlife model developed was the qualitative habitat matrix, which simply relates certain habitat types to the presence/absence of certain species. Such habitat "crosswalks" typically use either a binary classification (habitat/non-habitat) or a few classes (high/medium/low). Although simplistic, this approach is widely used because assessments attempt to include several species across large landscapes (Hulse et al. 2002; Johnson and O'Neil 2001).

Another early modeling approach focuses on population numbers rather than habitat. Such numerical population models predict future population numbers based on current population and simple reproductive and survival functions but without reference to available habitat. The Leslie matrix is one of the best known examples (Leslie 1945). One of the earliest management plans for the NSO was based on an aspatial population model (U.S. Department of Agriculture 1984). While more attention is now given to spatial habitat models, a few important population modeling tools continue to be available (e.g., see Possingham and Davies 1995; Lacy 2000; Applied Biomathematics 2011).

A second generation of NSO modeling incorporated meta-population theory by combining simple habitat assumptions with stochastic models of demography and dispersal. These models assumed a certain percentage of the landscape to be suitable habitat, which then influenced the success and survivorship of juvenile dispersing owls. Using a sensitivity analysis, Lande (1988) found that the probability of successful dispersal was the second in importance of six basic life history parameters, and integrated this factor into his model using unique terms for the

proportion of the region in suitable habitat, and the number of territories a dispersing juvenile might search. Doak (1989) further separated dispersal into within- and between-habitat cluster dispersal.

This increasing spatial awareness was integrated into a second major conservation strategy for the NSO, which called for larger habitat blocks (each large enough for 20 territories), spaced over the landscape, so that they were no more than 12 miles apart (Thomas et al. 1990). Little was known about dispersal needs, but the associated "50-11-40 rule" assumed that maintaining at least 50 % of the landscape between habitat blocks in trees $\geq 11''$ in diameter with ≥ 40 % canopy cover would be sufficient. The models stemming from this plan were the first to incorporate a specific spatial structure for the landscape into a dispersal model, although in an idealized fashion where the size and spacing of habitat clusters for each model run were uniform (Lamberson et al. 1994). A few environmental groups successfully blocked this "Thomas Strategy" in court, due largely to arguments over estimates of population demographics, but also involving the unrealistic modeling assumptions of a uniform landscape and maximally efficient dispersal patterns (Harrison et al. 1993).

Schumaker (1996) took another step towards realism by beginning with a raster map of binary predicted habitat (suitable/unsuitable) over the Northwest Forest Plan area (ROD 1994). He was also the first to test the applicability of more abstract landscape metrics by comparing them to results from a dispersal simulation. Schumaker's individual-based dispersal model has continued to evolve and be applied to a wide range of species. The first model version, PATCH, evolved into the current version called HexSim (Schumaker et al. 2004).

Akçakaya and Raphael (1998) also began with the Northwest Forest Plan habitat map, which they aggregated to 5.7 km^2 mapping units and identified areas of sufficient habitat to support one or more NSO home ranges. They applied the commercialized RAMAS GIS spatial meta-population software to NSO population modeling and modeled dispersal as a function of population size, distance, and patch size. Their research focus was on the effect of uncertainty in life history parameters, and they ran the RAMAS model using a range of life history inputs. In contrast to Lande (1988), they found that neither total NSO abundance nor population viability was sensitive to dispersal rate; however, survival was not integrated with dispersal rate.

While some NSO studies focused on population numbers, others have relied on habitat as a surrogate for population numbers. A further step in modeling complexity from simple habitat matrices includes additional detail on habitat features and more quantitative assessment. This sort of modeling, now widely used in wildlife habitat regulation and management, is referred to as habitat suitability or effectiveness modeling (USFWS 1981). Habitat suitability indices (HSI) score specific habitat features (e.g., number of trees $>20''$ diameter) on a common scale and combine scores into an overall suitability index. Individual indicators are typically combined via one of three types of functions: evaluating indicators in terms of a most limiting factor (minimum), as equal and compensating factors (average), or as cumulative factors (sum). Often indicators are grouped by the

species life history need they meet (e.g., nesting, roosting, or foraging habitat indicators). Habitat effectiveness models (or habitat evaluation procedures, as they are sometimes called) generally incorporate HSI-like habitat quality measures, along with habitat quantity measures to derive estimates or indices of population carrying capacity (Roloff et al. 2001).

HSIs can be developed from expert knowledge or derived empirically. An empirical approach, which has been recently applied to the NSO, is referred to as occupancy modeling; i.e., taking known nesting sites and determining which habitat factors can best predict them. Zabel et al. (2003) worked with a biological team to create six different habitat maps in northern California and 27 statistical model variations. They found that the official habitat map performed poorly, while the best model performance occurred at the core home range level (200 ha), combining different relationships between nesting/roosting and foraging habitat quantities. For the first major monitoring report on the Northwest Forest Plan (ROD 1994), Lint (2005) examined both NSO population trends and changes in habitat area. Nesting habitat suitability was predicted based on occupancy data using the Biomapper software (Hirzel et al. 2004). In addition to reporting habitat quantities by administrative units, they also calculated a number of landscape metrics using the Subdivision Analysis extension for ArcView (Lang 2004) and FRAGSTATS (McGarigal and Marks 1995) as part of the Patch Analyst extension for ArcView GIS (Rempel and Carr 2003). Available data were insufficient to model dispersal habitat suitability, so they used Thomas Strategy assumptions of 11-inch tree diameter and 40 % canopy cover.

As part of the Oregon Coastal Landscape Analysis and Modeling study, McComb et al. (2002) were first to explore possible effects of different future forest management options on NSO habitat in a spatially-realistic manner. Their HSI, constrained by variables in a forest projection model, consisted of tree densities in a variety of size classes. Their 25 m^2 mapping unit, considerably finer-grained than past efforts, employed a moving window approach, and tested seven possible models at four different scales (from 0.56 to 1810 ha).

Finally, in the most comprehensive spotted owl modeling effort to date, Sutherland et al. (2007) combined landscape modeling with multiple scales of habitat assessment and a population model for a study designed to support the Canadian Spotted Owl Recovery Team in British Columbia. They used the cell-based SELES landscape dynamics system (Fall and Fall 2001) to project alternative management scenarios, which were evaluated for NSO habitat using a multi-stage process. The habitat process began with a site-level HSI, which was then evaluated at a broader scale for viable owl territories. These territories were then analyzed for spatial connectivity using concepts from graph theory applied to the landscape of habitat suitability ratings (Urban and Keitt 2001). A meta-population model was then applied based on territories and connectivity, and outputs were combined within a Bayesian belief network to rank potential habitats.

2.2 EMDS Applications in This Problem Domain

Although this chapter describes the first application of EMDS (Reynolds et al. 2002) specifically for the NSO, it has been used in a variety of wildlife habitat-related projects. Table 1 summarizes a number of such modeling applications, which we discuss in more detail below.

One of the first applications to use EMDS assessed watershed condition under a newly established regional plan for monitoring federal forestlands in the Pacific Northwest, USA (Reeves et al. 2004). The monitoring program has used EMDS since 2001, which included a 10-year assessment of Northwest Forest Plan effectiveness (Gallo et al. 2005). These EMDS models have focused on aquatic habitat conditions for anadromous fish species (salmonids) because of their threatened status under the federal Endangered Species Act; however, models are also designed to evaluate aquatic and riparian conditions more generally. Because of this broad assessment objective and the inherent difficulties of developing statistical models for anadromous fish habitat, the program has relied on an expert workshop approach for model development and parameterization (Gordon and Gallo 2011). EMDS models have also been developed for other watershed assessment work, but from a regulatory rather than management perspective, including for the US Environmental Protection Agency (Reynolds et al. 2000), and for the state of California's North Coast Watershed Assessment Program (Dai 2004; Walker 2007). Extending assessments to prioritize watersheds for restoration, by incorporating feasibility and efficacy considerations, has been demonstrated as well (Reynolds and Peets 2001).

EMDS has also been used for fish and wildlife habitat evaluation as part of broader assessments. For example, Humphries et al. (2008) used EMDS to score the suitability of land for conservation reserves in the Interior Columbia Basin USA. Given the large planning area (58 million ha), they used a very coarse metric of habitat based on 35 vegetated cover types. Instead of evaluating specific habitat attributes (e.g., canopy cover), they considered disturbances, including road density, vegetation change, and disturbance regime changes. They also incorporated a basic spatial component by evaluating the size of each polygon, and where small, they integrated the scores of its neighbors. To derive optimized reserve networks, a measure of uncertainty was introduced by combining the EMDS polygon suitability scores with 15 potential acquisition cost scenarios using site-selection software.

White et al. (2005, and Chapter 9) used EMDS to identify conservation priorities in the 620,000 ha Sierra Nevada Checkerboard Initiative landscape (mixed public and private lands) west of Lake Tahoe, California, USA. For each 2.56 km^2 (1 mi^2) analysis unit, they assessed a large number of attributes ($\sim$50) pertaining to biodiversity, mature forest connectivity, recreation, and threats. For biodiversity, they combined coarse-filter (vegetation communities) and fine-filter (numbers of sensitive species) indicators for both aquatic and terrestrial habitats. Mature forest connectivity was evaluated using a variety of metrics calculated for each

Table 1 Key aspects of EMDS applications related to wildlife management

Application	Extent	Analysis units	Types of indicators	Criteria source(s)	Spatial metrics	Sensitivity/ Uncertainty
NWFP watershed condition (Gallo et al. 2005)	NW Forest Plan area (100,000 km^2)	Subwatersheds (40–120 km^2)	- watershed-level vegetation and roads - stream reach level physical and biological factors	Expert knowledge and interpretation of the literature	None	None
Desert tortoise (Heaton et al. 2008)	Army training center and surroundings (20,000 km^2)	Public land survey sections (2.59 km^2)	- habitat - population - threats	(Not specified)	Various proximity metrics	6 model structure alternatives
US Forest Indicators (Reynolds et al. 2008)	United States	Forest Service regions Ecoregions Counties States	Coarse filter changes in forest types to fine filter metrics of endangered species listings and population counts	- Mostly fairly recent history - A few historical reference conditions - Focus on trends	None	None
R1 National Forest Planning (Jensen et al. 2009)	3 Forest Service planning zones (580,000 km^2)	Ecological subsections (100–200 km^2) Subwatersheds (40-120 km^2) Counties	- habitat - population - threats	Relative to dataset (25/75th percentiles)	Patch size, interior habitat area	3 different prioritization objectives
Land conservation suitability (Humphries et al. 2008)	Interior Columbia River Basin (56,000 km^2)	17,227 biophysical units (3-6,943 km^2)	35 naturally vegetated cover types	Absolute cover type targets	Polygon size, neighborhood suitability	15 land cost scenarios
Sierra Nevada Checkerboard (White et al. 2005)	Central Sierra Nevada mixed ownership landscape (6,192 km^2)	Public land survey sections (2.59 km^2)	~50 pertaining to biodiversity, mature forest connectivity, recreation, and threats	Relative to dataset (Min/Max)	5 km buffer for neighborhood characteristics	None
Western OR High Conservation Value Forest (Staus et al. 2010)	BLM Western OR Plan Revision area (77,000 km^2)	Public land survey sections (average size of 2.13 km^2)	17 indicators pertaining to aquatic and terrestrial values, including spp. presence and vegetation characteristics	Absolute where available, relative otherwise (±1 SD)	Distance to nearest old growth patch Index of fragmentation	None

evaluation unit and its immediate vicinity, using a 5 km buffer. These themes also included inputs explicitly chosen to reflect their potential values in the future (e.g., current low development density indicating greater future conservation potential). Since the modeling objective was land prioritization for conservation, the majority of the evaluation criteria were set using the minima and maxima of each data set.

Staus et al. (2010) similarly looked at both aquatic and terrestrial habitats in a study to identify forest areas of high conservation value in western Oregon, which were not specifically managed for conservation. They used a number of fine-filter biologically-based indicators, such as presence of NSOs, murrelets and other rare species, as well as coarse-filter metrics of percent old-growth forest and spatial metrics of its distribution (fragmentation).

The Integrated Restoration and Protection Strategy of US Forest Service Region 1 (Jensen et al. 2009, Chap. 5) also included significant wildlife components. Similar to the Sierra Nevada study, they included topics for both aquatic and terrestrial biodiversity, each of which incorporated coarse- and fine-scale metrics of current condition and potential threats. Their analysis units varied by resource and data resolution, and vegetation and aquatic resource conditions were evaluated at the ecological subsection level (100–200 km^2, Bailey 1996) and subwatershed level (4,000–12,000 ha, Seaber et al. 1987, http://nhd.usgs.gov/), respectively. The assessment only needed a relative rating of areas, so it used the 25th and 75th percentiles of the data to set the lower and upper bounds on evaluation criteria.

EMDS has also been used to evaluate the condition of forest resources at the national level in the US. Evaluation included a number of indicators related to biodiversity. The 1992 United Nations Conference on Environment and Development (UNCED) produced a Statement of Forest Principles, which in turn has led to regional efforts to promote and harmonize reporting on national forest conditions. The US participates in the Montreal Process, Criteria and Indicators group, and produced its first major national report in 2004 (Guldin and Kaiser 2004). Nine indicators were grouped under a criterion for biodiversity. The US report generally attempted to quantify these indicators, but did not attempt to evaluate acceptable levels of each indicator. Reynolds et al. (2003) argued that such evaluation was an essential step, and demonstrated how this could be done using EMDS (see also Reynolds et al. 2008). The units of analysis varied with the indicators, which ranged from coarse-filter changes in forest types to fine-filter metrics of endangered species listings and population counts. Few reference conditions were available for setting evaluation criteria. In some cases, relatively recent historical data were used (1960s, 1970s) and the general strategy for the effort was to focus on trends starting from the initial report.

Most similar in scope to the single-species application detailed in this chapter, Heaton et al. (2008) used EMDS to identify the best translocation sites for desert tortoises (*Gopherus agassizii*) given a planned expansion of military training grounds in southern California, USA. In contrast to other wildlife habitat assessments, more is known about threats to the tortoise than about its biophysical habitat preferences, so many of the indicators used in the model represented threats (e.g., roads, urbanization) and opportunities (i.e., land ownership). Evaluation

criteria were assigned by an expert panel and weighted by grouping them into two tiers; a top tier combined criteria with an AND operator (limiting factor approach) and a lower tier combined with a UNION (averaging) operator. Many of the indicators were spatial in nature, involving proximity to features (roads, urban areas). The analysis did not involve any measure of landscape configuration (i.e., landscape metrics), but the team made an especially effective use of alternative scenarios to incorporate differing scientific opinions. They constructed seven scenarios, with each altering the use of one or more indicators under contention. The results of all scenarios were overlaid and areas with high suitability values in every scenario were identified as the preferred translocation sites.

From the examples above, we clearly show that EMDS has been used successfully for a number of wildlife-related studies, usually dealing with large landscapes (>5000 km^2). Similarly, analysis units used in evaluations have been large (>2 km^2) and often based on ownership/management divisions than on ecological considerations. Indicators used have ranged from exclusively coarse-filter cover types to fine-filter metrics related to individual species, and often both have been combined in a single model. While EMDS is not designed to have any population modeling capabilities, population numbers and species presence/absence have been incorporated into a number of the assessments. Nor does EMDS have built-in spatial analyses, yet a number of studies have incorporated spatial proximity and fragmentation metrics through preprocessing, and assigned these values to individual analysis units.

Finding published, objective criteria for use in data evaluation appears to be a challenging modeling aspect. Many studies simply used reference statistics from the datasets themselves (e.g., minimum/maximum, 25th/75th percentiles), which provide a relative comparison but say little about habitat values in any absolute sense. Some studies applied more externally-relevant criteria, generated from analyses of historical conditions or drawing on established habitat values. By and large, uncertainty/sensitivity was not addressed in these studies. Jensen et al. (2009), however, demonstrated how model structures could be reconfigured to address different management objectives (management uncertainty), and Humphries et al. (2008) combined their outputs with 15 land-cost scenarios to evaluate sensitivity to these costs. Heaton et al. (2008) went the farthest in addressing uncertainty in their outputs by running five alternative model formulations and then prioritizing areas scoring highly on all five.

3 Motivation for Using EMDS in This Context

The DNR's principal need for the South Puget NSO analysis was to rate dispersal habitat conditions across a set of management alternatives. This focus meant that a habitat rather than a population model was the most appropriate approach, and, since little data were available on the needs of dispersing owls, the model would need to be largely informed by expert opinion. The DNR informally reviewed a

number of approaches and chose EMDS for a number of reasons. EMDS had been used in a number of habitat assessments already (as described above), and its basis in fuzzy logic provided a mechanism well-suited to the integration of expert knowledge. The fuzzy logic approach also provided a simple solution to problems identified with the strict threshold approach to habitat definition; EMDS provides a method to integrate a variety of indicators and can produce combined evaluations on a continuous scale. Finally, the polygon-based framework used by EMDS fit well with the DNR's stand-based forest inventory data, and with the division of these stands into finer-grained analysis polygons.

4 What Worked Well

4.1 Data Structure

The first step our project team took was to develop a prototype model based on DNR's current inventory data. These data were organized by stand polygons, so it was quite easy to link the data into an EMDS model. There were also a considerable number of stands lacking certain indicators or inventory data altogether. EMDS accommodates such data gaps by assigning them an evaluated score of zero (no evidence for or against the proposition). This feature allowed the team to produce a prototype model quickly, which in turn helped team members and owl experts to understand the approach and see its advantages. The fact that EMDS is scale independent was also a significant advantage, since, for the forest modeling alternatives, the original stand data were further subdivided by intersecting a number of features relevant to the planning process, such as stream and road buffers.

4.2 Expert Engagement

Wildlife models designed to answer management questions must often rely on wildlife experts to synthesize available information to provide the model structure and key parameters. The processes used to engage experts can involve significant time and effort, but they positively influence model quality (Andelman et al. 2001; Gordon and Gallo 2011). Here, we describe our process and experiences based on detailed notes from a dedicated note-taker and a written post-workshop review compiled by the project team.

The project benefitted from a number of previously conducted reviews of NSO habitat characteristics (Courtney et al. 2004; Hanson et al. 1993; Thomas et al. 1990); however, as mentioned above, there was little empirical data on the habitat needs of dispersing owls. DNR managers recognized that there was a significant effort needed to synthesize the available data in a manner that took advantage of

EMDS modeling capabilities. To do this synthesis, the project team identified a group of NSO experts: two biologists from the DNR (in addition to the project lead), and one each from the Washington Department of Fish and Wildlife and Oregon State University. To quickly produce a model, which would go out for broader review as part of the draft forest plan, a small group of experts most immediately involved in the state forest planning process was chosen.

We scheduled a two-day workshop to begin the expert engagement process. Given the limited amount of time available, the team prepared for the workshop carefully, including the development of a written plan comprising the following elements: equipment, materials (presentations, handouts, computer reference files), roles (manager, facilitator, data provider, and note-taker), and a detailed agenda of modeling phases. 2 weeks prior to the workshop, we provided a packet of background materials on the EMDS modeling process, the available data, and a set of indicator values extracted from the literature. The availability of a science synthesis report (Hanson et al. 1993), prepared for the 1997 HCP, provided a strong base for beginning discussions of the habitat model.

We began the workshop with three presentations on the background and objectives, EMDS modeling process, and available data. We then moved into model specification, using the following order to proceed from overall model structure to the individual indicators of owl dispersal habitat (steps are described below):

1. Sketch out model structures
2. Choose the specific indicators
3. Choose evaluation (scaling) criteria for each indicator
4. Document follow-up tasks and concerns.

The experts appeared to have little difficulty understanding the EMDS approach to modeling, and the choice of indicators proceeded rapidly. Instead of building one habitat model, they choose to build three, in order to reflect different life history requirements of dispersing owls: foraging, roosting, and movement. This division was somewhat unusual for a habitat suitability model, and it was important in recognizing that not all portions of the landscape must meet all functions in order to support dispersal. Conceptual diagrams of the models are presented in Fig. 2a–c. Model structures were helpful in deconstructing individual life history requirements into indicators, such as thermoregulation. At the finest level of disaggregation, indicators had to be directly or indirectly measurable in the field. For example, while thermoregulation itself is difficult to measure, indirect indicators that provide thermoregulatory habitat (e.g., canopy cover) can be quantified by taking measurements in forested stands. The process slowed when we reached the third stage: setting the evaluation criteria values for each indicator (which we discuss further in the "Limitations" section). This slowing was understandable, since it represented the operational decisions regarding numerical threshold values of some significance, and the most appropriate datasets to use. Not all aspects of our model were specified in a single workshop—questions

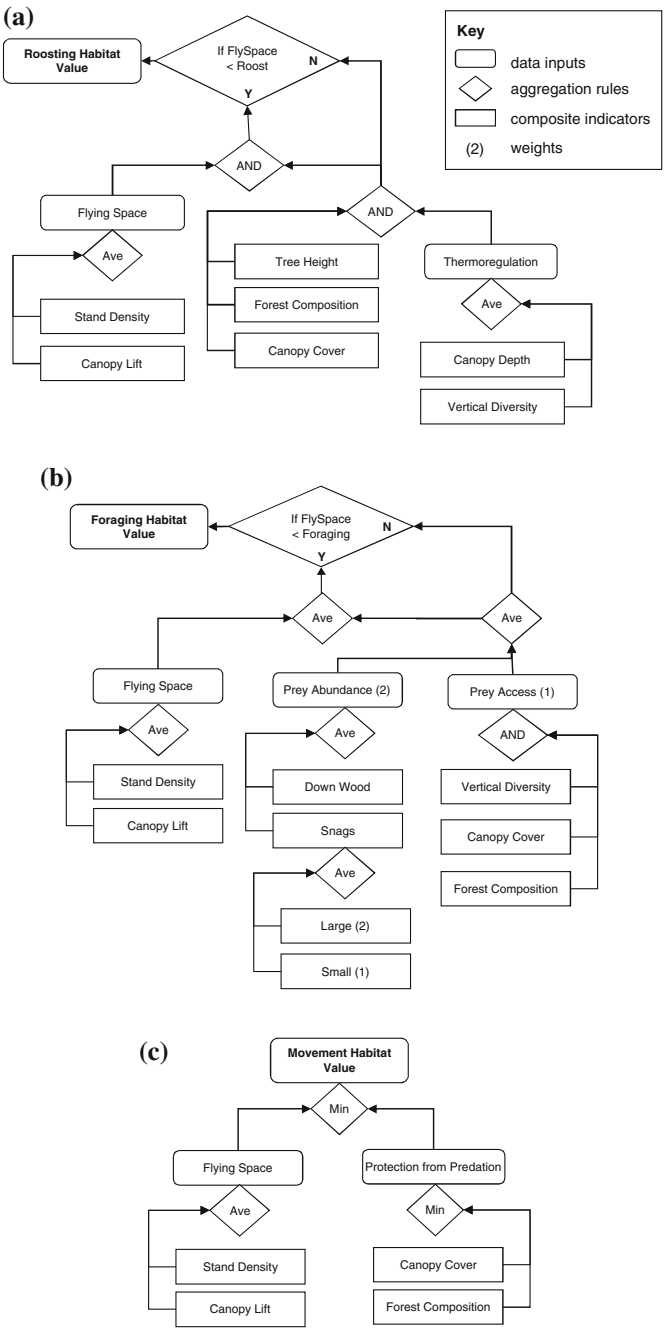


Fig. 2 EMDS model structure diagrams. **a** roosting habitat model; **b** foraging habitat model; **c** movement habitat model

requiring further research often arose. These were recorded in a special section of the model—Notes for Follow-up.

4.3 Data Evaluation and Integration

One of DNR's major objectives was to move away from a simple binary approach to habitat management, where each indicator either passes or fails. It was frustrating to DNR managers that this pass/fail system would not distinguish a recent clearcut from a stand approaching (but not quite meeting) threshold conditions for dispersal habitat. The fuzzy-logic in EMDS provided a solution to this problem. Once evaluation criteria were chosen, each indicator was scored on a standardized, continuous scale; stands closer to optimum conditions received higher scores.

Developing evaluation criteria is often the most challenging aspect of EMDS model development. For example, in the case of the NSO, a key question centers on the tree diameter threshold that differentiates "no support" from "support" for good habitat. A number of EMDS applications discussed above were able to solve this problem by basing the criteria on the distribution of the data being analyzed (e.g., all stands >80th percentile are scored +1). However, a relativizing approach assumes that either desirable conditions can be captured by a known percentile of the current conditions, or that only a relative comparison is needed. Since the DNR wished to reassess the very definition of dispersal habitat, it was unknown what percentile of current conditions was acceptable. Although the ultimate use of the spotted owl evaluation was in fact a relative comparison between management options, relative criteria (i.e., chosen simply based on statistics, such as maximum and minimum) were insufficient for a few reasons. More absolute criteria (i.e., based on external considerations, such as actual habitat needs) had already been established by the HCP and mapped, and any new habitat assessment method would be viewed with a similar expectation. The HCP required certain amounts of habitat to be produced, rather than just choosing from among the best options.

Since single indicator evaluation thresholds had already been established by the HCP, the project team started with these indicators in the development of their prototype. The binary threshold of each indicator was spread into an upper and lower bound by applying a percentage multiplier, e.g., the 11-inch tree diameter was multiplied by ±20 % resulting in lower and upper bounds of this criterion of 8.8″ (−1, evaluates to no support) and 13.2″ (+1, evaluates to full support). This approach enabled the quick generation of a prototype, but the DNR saw the need to have these thresholds reviewed and updated by an expert group. The experts appeared to readily understand and appreciate the fuzzy logic approach to indicator evaluation.

Another shortcoming of HCP binary habitat definitions was that any stand could fail to meet habitat conditions by narrowly missing a threshold for a single indicator. The EMDS approach of producing a standardized score for each indicator, and aggregating scores within a hierarchical (or network) structure, provided a

solution to this problem. In some cases, the owl experts maintained a limiting factor approach similar to the HCP (a MINIMUM aggregation function), but in others they allowed partial compensation between indicators (AVERAGE operator). Indicators were also weighted in a few cases. Interestingly, one of the first steps taken by the experts was to divide the habitat assessment into distinct models for three different life history needs of owls: roosting, foraging, and movement. It was felt that previous thresholds had been facilitated safe movement of owls through the forest, without considering concurrent roosting and foraging needs. The models generally contained one intermediate level of aggregation based on more detailed aspects of life history needs ("Prey Abundance" in Fig. 2). As mentioned above, the experts did not appear to have much difficulty in devising the hierarchies for the three models.

Initial results from the models produced a pattern counterintuitive to the project lead's expectations: the movement model, which was expected to be less demanding, rated more stands poorly than either the roosting or foraging models. By comparing the models, we determined that the lower scores were caused chiefly by a Flying Space node (a combination of stand density and canopy lift), which was in the movement model, but not in the others. The expert group felt that flying space was an important consideration for foraging and roosting also, and the modularity of the EMDS model enabled us to quickly copy this structure into the other models.

4.4 Model Validation

Model validation as usually understood in the natural sciences means testing to see if a model produces empirically accurate results with respect to independent, real world observations (Oreskes et al. 1994). However, knowledge-based systems are often built for situations in which such empirical tests are neither possible nor affordable. For this reason, validation in the expert systems sense is often done by comparing model processes and results back to the judgments of experts (Turban and Aronson 2001). Sufficiently detailed data on habitat use during dispersal were not available, so our validation strategy was limited to consideration and confirmation of model results by the expert panel.

Our first step was to send out the model results to the expert group via email. We received some feedback via email, but it was piecemeal and driven by a few initial questions from one of the participants. To get a more comprehensive assessment, we scheduled a follow-up model review workshop. Because of a difficulty experts had expressed in visualizing the stand-level inventory data during the first workshop, we decided to spend the first day of the review workshop in the field, examining a few sample stands and their indicator scores, and comparing them with associated data values.

We tested a number of handouts designed to enhance the visit. The first was simply an aerial photo of the stand, which provided the big-picture view, since we

would only be visiting one point in the larger stand. Second, we provided a stand statistics page with the indicator values from the stand inventory. We also included a graphic picture of the stand produced with the Stand Visualization System application (McGaughey 1997), to test how well such visualizations might compare with actual conditions. Third, we provided a "visual stand scoring sheet," on which we asked each expert to independently score stand indicators based on their visual assessment after observing each stand. Experts were then asked to share their scores, and discussions ensued about the numeric evaluation points for each indicator. Finally, we prepared a set of questions to ask at each stand concerning the most problematic indicators. The experts found the handouts quite useful. The visual scoring sheets and prepared questions helped keep the conversations on topic, but we were not ultimately able to collect enough scores to compare to the EMDS result scores.

A few months after the second workshop, we sent the revised stand model results out by email for a third round of review, along with a first draft of results from the landscape model. The expert group had a few remaining questions on the stand model but seemed largely satisfied with the results.

5 What Didn't Work Well

5.1 Data Structure

Our EMDS model adapted well to the DNR data structure; however, managing the fourth dimension (time) was more challenging. The environmental impact statement (EIS) process involved estimating the impacts of three alternative management options over the next 100 years. A 10-year time-step was chosen by the EIS team, which meant we would process 30 (3 alternatives × 10 periods) data tables as input to the owl model. Given that multiple runs were needed to test and refine the operation of the model, this represented a considerable analytical burden. Initially, we reduced the load by simply limiting our analysis to three of the 10 periods (beginning, middle, and end). Ultimately, however, we wished to see the full trend over time.

ArcMap and EMDS did not have a facility for handling a time series of data by assigning more than one value to a single polygon, so our initial approach was non-spatial. We developed Python® (http://www.python.org/)code to stack the 30 input tables into one master table, with new fields to track which "alternative" and "time period" each record was associated with. Then we ran this combined table through the EMDS-NetWeaver module directly, which does not require spatial objects to be associated with the data.

This approach worked reasonably well for the stand-level model done in the EMDS framework. However, as our landscape model evolved (described further below in Data Evaluation), we began to see the potential advantages of integrating the two scales. Without such integration, a model run involved multiple human

interventions: an initial preparation of the data for NetWeaver, the NetWeaver run, and finally the initiation of the landscape model run. During the draft EIS review process, after comments were tendered requesting additional model runs to test the sensitivity of various parameters, we decided to fully integrate the stand and landscape models. We replicated the functionality of the basic NetWeaver fuzzy logic operators in a Python script, so that the complete model and all iterations could be performed as a single program run. The basic functionality was not particularly difficult to mimic: nevertheless, the integration and testing required considerable time of an expert programmer over several weeks. Another advantage of this approach was that it could be more closely integrated with the DNR data structures, drawing on and writing to them directly without manual intervention. Disadvantages were the loss of other EMDS functionality (e.g., quick creation of output maps and scenarios) and the programming expertise necessary to maintain the model.

5.2 Expert Engagement

As described above, the owl experts expressed difficulty in visualizing stand conditions from the statistics provided (e.g., trees per acre). They referred to stand photos shown in one of the introductory presentations, which suggested the need for additional photos representative of stand conditions and classification of stands into a few basic types. They also requested seeing frequency distributions of the indicators to aid them in setting thresholds. We were able to provide some displays "on the fly," but preparation of a handout would be more helpful in the future. In our post-workshop review, we also agreed that beginning the assessment of each indicator by explicitly reviewing our literature summary (from the background materials) would have been helpful.

In general the experts appeared to quickly understand the EMDS modeling framework. Where we encountered the most difficulty was in sharing the modeling results with the group as part of the validation process. We could not expect the participants to run GIS or other parts of the modeling software, and thus needed to encapsulate the results in a more accessible format. Simply exporting the EMDS result tables to a spreadsheet produces a mass of variable names and numbers that were not ordered according to the model hierarchy. We worked around this by using database software (Microsoft Access) to reformat and export the results to a spreadsheet format (Microsoft Excel). Additional manual formatting and writing of explanatory text was also done. We manually exported EMDS map results and imported these into PowerPoint. Through these steps we were able to make the results accessible; however, the EMDS modeling process would have greatly expanded utility if these needs could be better addressed by the software.

Our second major problem with expert engagement was not due to, but rather highlights, the utility of EMDS. The second day of the initial expert workshop focused on modeling landscape connectivity. We presented only a few general

ideas on the various scales that might be relevant to this task (e.g., neighborhood, patch, landscape), but the value of a chosen analytical framework, such as EMDS, became quickly evident, because with no specific roadmap to follow, the resulting discussion was unfocused and alternated between broad biological principles and narrow computational challenges. A number of these broad principles were eventually useful to frame the EMDS model (e.g., risk is determined by the size of roosting/foraging patches and the connectivity between them, owl dispersal needs vary between transience and colonization stages), but little overall progress was made in assembling a functional model specification.

5.3 Data Evaluation and Integration

EMDS worked well for the basic stand-level habitat assessment. The experts appeared to readily understand and appreciate the fuzzy logic approach to indicator evaluation; however, based on progress made during the initial workshop, choosing the evaluation criteria values seemed considerably more difficult than choosing the indicators and structuring them into a model. This is not a limitation of EMDS, but a limitation of expert knowledge, which EMDS exposes.

The fact that we were looking outside the EMDS framework for a solution to the landscape connectivity question does indicate one of its limitations. A number of other EMDS models have integrated spatially-aware attributes: Bourgeron et al. (2000) used polygon size and the suitability of neighboring units; White et al. (2005) also used neighborhood suitability scores; and Heaton et al. (2008) used proximity to other spatial attributes, such as roads. All of these metrics were based on the proximity of attributes to each polygon. Although EMDS does not perform this "windowing" itself, it is easy enough since it sits inside a GIS application and the results are directly assignable to the individual members of the existing polygon set. In contrast, our overarching question was the connectivity of the landscape as a whole, which required a more holistic assessment.

We ultimately settled on a graph theoretic approach (Urban and Keitt 2001), based on identifying habitat patches and the distances between them, which we then summarized using landscape metrics (Saura and Pascual-Hortal 2007). Constructing the patches required combining the EMDS model scores from the roosting and foraging models (both had to meet a minimum threshold) and then aggregating these polygons from the EMDS stand-level assessment. In using a patch-based approach we lost some of the finer gradation of scores produced by the fuzzy logic, because an in/out patch threshold had to be chosen, but we were able to preserve some of this information by assigning a patch suitability score based on size and quality.

Whereas previous owl modeling efforts generally have assumed that habitat between patches is uniform, our focus on dispersal led us to give this aspect more thought. Cost distance is a well-established technique in GIS, in which the distance from one point to another is calculated, based not only on distance, but also on the

difficulty of traversing the intermediate terrain. A number of researchers had applied it to animal habitats (e.g., see Bunn et al. 2000; Singleton et al. 2002). Problems with applying cost distance approaches to habitat assessment include broadening its application to patches instead of points, and working around its GIS implementation, which focuses on finding the least-cost path, an unrealistic assumption for species without access to GIS. We based our model on recent work by Theobald et al. (2006), which addresses both of these issues. Cost distance is generally implemented using a raster rather than vector data format, which required us to convert the data to raster as it was moved from the stand to landscape sections of the model. Adding the capability of handling raster data to EMDS would increase its applicability to a wider set of analyses.

5.4 Model Validation

A few issues related to model validation have already been discussed above: providing the model output in an accessible format and keeping the expert team engaged enough to understand the technical aspects of the model. With the publication of the draft EIS, the validation emphasis shifted focus from the expert group to the broader public. In terms of the validity of the model to assess owl habitat, the only major comment received actually came from a member of our expert panel, but this was more a reflection of his role as the representative of the state wildlife regulatory agency. The comment centered on the need to acknowledge and incorporate uncertainties into the analysis.

Discussions of uncertainty in the fuzzy logic literature tend to focus on linguistic uncertainty—how it addresses language concepts such as high, medium and low (Adriaenssens et al. 2004; Regan et al. 2002). We found little explanation of how such systems address uncertainties in model inputs and relationships, which was the focus of the comment. Such uncertainties seem more easily explained in the use of Bayesian belief networks, which are often used for similar applications, because they explicitly combine probability distributions. However, these distributions tend to be greatly simplified, often into bins such as high, medium and low. Uncertainty in the partition points for these bins resembles that of inflection points on fuzzy logic functions, but the latter appear harder to explain simply using the language of certainty because they are tied to fuzzy set theory instead of probability theory. However, it could be argued that fuzzy logic naturally incorporates a degree of uncertainty in inputs, which is reflected in the shape of the fuzzy evaluation functions.

Because we did not find sufficient discussion about input uncertainties in the literature and the concept is not intuitively familiar, we employed a standard sensitivity analysis approach. We discussed the perceived degree of uncertainty around each of the model parameter inputs with the regulatory agency representative and chose alternate plausible values for four aspects perceived to be the most uncertain. Three of these aspects were related to parameters in the stand model

(down wood, snags, and their combination) and one was in the landscape model (the costs of moving through unfavorable cover types). We re-ran the model for each of these four different assumptions, which resulted in a range of final landscape scores that varied from our baseline assumptions by −26 to +41 % in the initial period to −26 to +15 % in the final planning period (WADNR 2010a).

6 Conclusions

EMDS has been shown to be an effective tool in the wildlife modeling domain. Its primary uses have been in habitat suitability assessment. Its basis in fuzzy logic is compatible with techniques developed for habitat suitability indexing when using ramp functions to evaluate specific habitat attributes and when combining these attribute scores using common mathematical operators to obtain overall suitability scores. EMDS is not a simulation system, so it is not suitable for population simulations; however, some applications have successfully integrated population estimates (from other sources) into EMDS assessments and prioritization models. In the same vein, EMDS does not include spatial neighborhood functions, but a number of cases have included such metrics in assessments by pre-calculating them in a GIS. In sum, the EMDS framework is most appropriate for species dependent on habitat variables that can be mapped to specific spatial units; it is less suitable for analyzing influences based on population size or spatial behaviors, such as herding or migration. However, the simplicity and flexibility of the EMDS framework have led to its use in large landscape assessments that attempt to integrate diverse factors (e.g., habitat, population, threats).

Our analysis of NSO dispersal habitat benefitted greatly from the analytical framework provided by EMDS. The software enabled the rapid development of a prototype model, and it facilitated the use of expert knowledge to define habitat for dispersing NSOs. We did encounter limitations though, especially related to the spatial modeling of owl dispersal and the automation of multiple model runs.

Many analyses of wildlife habitat in forested regions begin with polygon-based maps from forest stand inventories. The EMDS polygon-based data structure fits this type of data well, and can easily manage polygon subdivision produced by combining data layers. However, many wildlife analyses also use raster-based functions, which are generally superior for analyzing proximity and connectivity issues. Some EMDS applications have worked around this by using regular grids of polygons (Heaton et al. 2008; White et al. 2005), but this approach does not provide access to many raster-based analysis functions in GIS software. Dispersal modeling places a special emphasis on habitat connectivity, and we found that this aspect of the analysis was better done in a raster environment. The ability to work with raster data would be a particularly useful addition for wildlife modeling with EMDS. Integrating EMDS with subsequent raster analysis was also a challenge, which may be solved by exposing some of the basic EMDS analytical functions as tools within the ArcGIS environment. Exposing such functions could further help

with the challenge of automating model runs for multiple time periods or sensitivity analyses.

The fuzzy logic engine in EMDS was effective at moving habitat scoring from a strict threshold-based system to a finer, multi-valent gradation of habitat values. This change in itself was quite significant in the eyes of many DNR staff, who had long been frustrated with the insensitive, black and white definition of habitats specified in the HCP. Fuzzy logic also provided a well-accepted method for using expert knowledge to set habitat evaluation criteria. This aspect was crucial given the lack of direct empirical data on the habitat needs of dispersing owls—a situation which occurs all too frequently in wildlife modeling. The new polygon-based habitat maps were a useful product themselves. Such maps were a familiar format for DNR staff and the EMDS methodology was relatively easy to explain. Although the EMDS framework appears easy to grasp, we believe the software could benefit from including functions to help package and share the results with experts, managers, and other stakeholders who cannot be expected to run the program themselves.

Finally, quantifying model uncertainties appears to be increasingly important to model users. In addition to our work, at least two other EMDS wildlife analyses have addressed uncertainty through the use of scenarios (Bourgeron et al. 2000; Heaton et al. 2008). Such analyses have simply involved altering input parameters or model structure and re-running the model. The EMDS scenario capability provides good support for this type of sensitivity analysis, but support could be improved by providing more automation tools to run a range of values in batch mode. More fundamental discussion and analysis in the literature of how fuzzy logic manages uncertainty would also be useful for analysts and stakeholders.

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