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Experts and models can agree on species sensitivity values for conservation assessments

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ABSTRACT

Species sensitivity values can be used to trigger management interventions and prioritize areas for conservation, with sensitivity estimation methods ranging from expert opinion to empirical modelling. The opinion and modelling approaches each have strengths and weaknesses, raising questions of how much they (dis)agree or which one to follow in conservation assessments. We compared conservatism values assigned by botanists to modelling estimates of sensitivity (change in abundance between current and reference conditions) for 123 wetland macrophyte species across northern prairie and boreal forest regions of Alberta, Canada. Scores from each method were positively correlated and showed limited differences especially in the boreal region. Conservatism distributions for species were broadly similar between regions whereas model-based score distributions differed between regions, probably because the modelling incorporated site-specific responses of species to environmental conditions prevalent in each region. A few species had large mismatch between conservatism and model-based scores, but these cases resulted from extenuating factors and do not reflect systematic bias in expert opinions or the modelling process. Overall the results indicate potential for general agreement between quantitative and qualitative methods of sensitivity estimation, and a complementary approach of expert opinion and modelling may offer the most valuable currency for conservation assessments.

1. Introduction

Sensitivity of species to anthropogenic disturbance is commonly used to assess ecological condition and degradation, or to trigger management interventions and prioritize areas for conservation. Typically, the degradation or status of an area is inferred against some benchmark in time (pre-European settlement, time-zero) or in areas considered minimally or least impacted by humans (Reynoldson et al., 1997; Bailey et al., 2004; Stoddard et al., 2006; Hawkins et al., 2010). The methodology can range from being fully objective and empirical, such as predictive models of taxonomic completeness (Hawkins and Carlisle, 2001), to the pure subjectivity of expert opinion (Lamb et al., 2009). Expert opinions and predictive models can each have strengths and weaknesses, which has raised questions of how much they (dis) agree, which approach to take, or whether to integrate them in conservation practice (Cowling et al., 2003).

Species sensitivity can be quantified based on how an ecological variable (e.g. abundance) observed under current environmental (natural and anthropogenic) conditions compares to that observed under minimal human activity. Nielsen et al. (2007) offer a good example of this empirical approach to sensitivity (they used "intactness") estimation. Using high-resolution geospatial layers, extensive field-collected data, and species-environment modelling, predicted species abundances are compared between the current mosaic of natural land cover and human footprint (e.g. agriculture, urban/industrial development), and the estimated land cover that previously existed within that footprint (ABMI, 2016). Species sensitivity is then measured as the deviation of its predicted abundance under the current conditions from that estimated under pre-footprint conditions (Nielsen et al., 2007). The site-specific deviation values for species can be averaged to estimate sensitivity at broader taxonomic levels over a given area or region (ABMI, 2016). This method may provide a more statistically robust and ecologically relevant index of conservation status (i.e. departure from minimally altered conditions) than traditional measures of biodiversity change such as species richness (Fleishman et al., 2006; Lamb et al., 2009; Hillebrand et al., 2018). However, the data demands can be prohibitive, especially when dealing with rare or elusive species, and any modelling process can have biases and technical difficulties.

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Alternatively, species sensitivity can be based entirely on informed professional opinion. A good example is the species conservatism values available for vascular floras throughout much of the United States and parts of Canada (Bried et al., 2012; Wilson et al., 2013; Freyman et al., 2016). Field botanists designate coefficients of conservatism based on their knowledge of regional floristics and species distributions and, to a lesser extent, life history. The scores are intended to reflect the probability of the species' occurrence under remnant or minimally altered conditions and its relative tolerance to anthropogenic disturbance (Taft et al., 1997). They can be averaged to estimate the overall floristic conservation value of a sampled area for purposes of prioritization, restoration monitoring, or compensatory mitigation (Spyreas and Matthews, 2006; DeBerry et al., 2015). Mean conservatism can provide a stronger biological condition assessment than species richness and other standard biodiversity measures (Taft et al., 2006). The main criticism, at least historically, is the subjectivity of conservatism assignments, but this concern has always lacked empirical support (Chamberlain and Ingram, 2012) and increasingly appears unwarranted (Matthews et al., 2015; Bauer et al., 2017; Mabry et al., 2018).

The opinion and modelling approaches are quite different yet may serve the same purpose in conservation monitoring and assessment. If the sensitivity values derived from each method generally agree, they may be used interchangeably or as complementary measures for improved monitoring and assessment. In contrast, systematic disagreement may indicate bias or error with at least one set of values, requiring further exploration to choose the best approach. We compared the conservatism values and modelling estimates of sensitivity for wet meadow vegetation in the northern prairie and boreal forest regions of Alberta, Canada. Expert knowledge is commonly used to inform modelling in conservation and applied ecology (e.g. Perera et al., 2012), but to our knowledge, no previous studies have looked at the level of discrepancy between opinion and modelling in the context of sensitivity-based monitoring and assessment critical to understanding biological degradation, management progress, and conservation status.

2. Methods

2.1. Study location

Our study area encompasses the northern prairie (hereafter “Parkland”) and boreal forest (hereafter “Boreal”) transition in the western Canada province of Alberta (Fig. 1). The Parkland region covers 9% of Alberta and marks the northern extreme of the North American Great Plains. This region is characterized by a mosaic of forest patches, extensive cultivated areas, and seasonal and permanent wetlands interspersed with urban and industrial development. It also has the densest human population in Alberta. Elevations in the Parkland range from about 300 m to 1500 m a.s.l. The Boreal region is much larger (Fig. 1), covering almost 60% of Alberta. It is characterized by flat to gently rolling plains (elevation range from about 150 m to 1100 m) dominated by vast forests and wetlands, mainly peatlands (bogs, fens) interspersed with areas of non-peat marshland and shallow open water. Although localised areas of the Boreal have been significantly altered by intensive forestry and energy development, large tracts of the region remain relatively undisturbed. Both regions experience short, warm summers and long, cold winters although the Parkland has higher mean annual temperatures (NRC, 2006; ABMI, 2017a).

2.2. Opinion method

We used conservatism values assigned to marsh and wet meadow plant species in the Boreal and Parkland regions (Forrest, 2010; Wilson et al., 2013). Nine botanists from academia, government, and consulting were contracted to assign region-specific conservatism scores to 407 species commonly associated with shallow emergent marsh and wet meadow habitat (wooded or graminoid) in both regions. These species were selected from previous Boreal and Parkland wetland studies and the Alberta Natural Heritage Information Centre as typical of marshes and wet meadows in Alberta (Vujnovic and Gould, 2002; Forrest, 2010).

Scores ranged from 0 to 10 with 0 assigned exclusively to non-native species; 1–3 to native species found with a variety of plant community types and appearing relatively tolerant of human disturbance; 4–6 to native species usually associated with a particular plant community and appearing to tolerate moderate disturbance; 7–8 to native species associated with a particular plant community and appearing sensitive to moderate disturbance; 9–10 to native species appearing sensitive to any disturbance (Forrest, 2010; Wilson et al., 2013). Botanists gave scores only when they felt sufficiently knowledgeable about the species, and in most cases strong disagreements among botanists were resolved before taking the integer-rounded median score across botanists (Tables 1, S1).

2.3. Empirical method

The model-based sensitivity (Nielsen et al., 2007) combines field sampling and geospatial data for 508 Boreal and 78 Parkland wetland sites (Fig. 1). These sites were sampled on a single-day July visit in one or two years during 2007–2016 following a standardized protocol (ABMI, 2017b). Sites were stratified into open water, emergent, and wet meadow zones with all vascular species identified in a max of three 10 × 2 m plots per zone (see ABMI, 2017b for details). Species were identified in the field and by plant taxonomists at the Royal Alberta Museum; taxonomy follows the Flora of North America (FNA, 1993 +) and the Integrated Taxonomic Information System (available at: http://www.itis.gov). We analyzed data from the wet meadow zone to reduce within-wetland natural variability, choosing this zone because (1) it supports more plant species than the open water and emergent zones which helped facilitate our analyses, and (2) previous work in the region found that the wet meadow zone had higher sensitivity to disturbance than all other wetland zones, and that combining plant communities across the elevation gradient resulted in a reduced signal (Wilson and Bayley, 2012).

We modelled the relative abundance (% of plots occupied) of species with at least 20 site occurrences using spatial coordinates, physical-chemical covariates, area of surrounding (250 m buffer) vegetation/soils and human footprint, and bioclimatic variables (see Table 2). The physicochemical covariates came from the field sampling (ABMI, 2017b) and buffer zone covariates from data layers produced by the ABMI Geospatial Centre. Instrument-collected bioclimatic data (4-km resolution) were spatially interpolated and averaged over 1961–1990 (Hijmans et al., 2005).

For each species we used a multi-stage model selection and averaging approach to predict abundance at each site in relation to surrounding land use (human footprint) and natural environmental heterogeneity (ABMI, 2016, 2017c). First, we fit a series of binomial generalized linear models *within* the physical-chemical, surrounding vegetation/soil, and bioclimatic-spatial sets of covariates (Table 2), retaining the best model ($\Delta_i = 0$) from each covariate set according to Bayesian information criteria (BIC). We then compared those retained models to select (using BIC) a final model that best captured the species relationship to natural (i.e. non-footprint) environmental heterogeneity. This model and two human footprint models (~total human footprint, ~alienating + successional footprint; see Table 2 for definitions) were used for abundance predictions. The estimated coefficients from these models were combined into a single abundance prediction using model-weighted averaging (Burnham and Anderson, 2002).

We created two site-specific predictions for each species' abundance using the model-averaged coefficients: one prediction under current conditions with all covariates including the most recent (circa 2014) human footprint inventory, and another under reference conditions where the footprint was backfilled with the natural vegetation cover