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

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A B S T R A C T

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

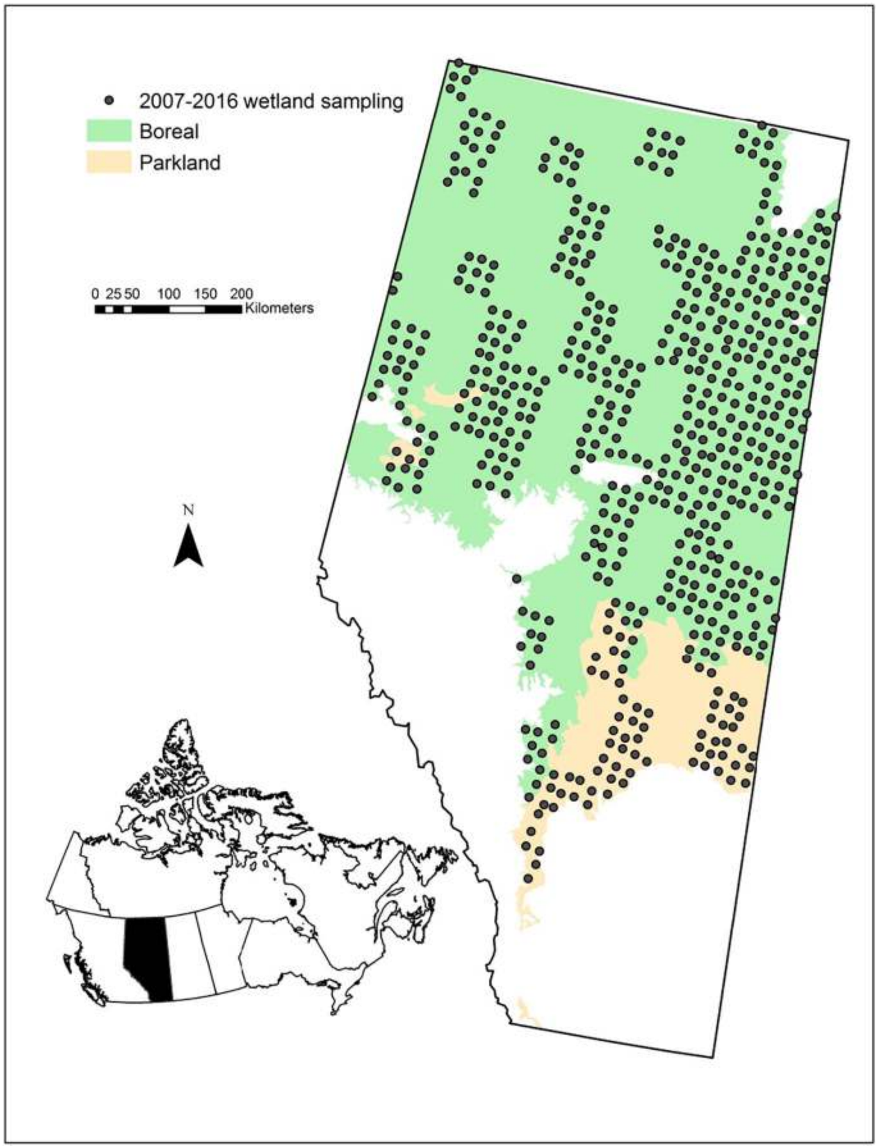

Fig. 1. Wetland sample points used for sensitivity modelling in two regions of Alberta, Canada. Each site was sampled on a single-day July visit in one or two years over the course of a decade.

surrounding each wetland (ABMI, 2017d). In brief, the backfilling process involves rule sets to select human-disturbed polygons divided into four categories: linear human footprint (e.g. roads, railways, pipelines), forest cutblocks, peat extraction, or other. These polygon types get backfilled according to category-specific procedures and rule sets, running a "multipart to single-part" GIS operation to ensure unique attribution of the natural vegetation and correct size of each polygon (see ABMI, 2017d for details). We defined each species' regional sensitivity as a ratio of the predicted abundance (summed across the wetlands within each region) under current conditions to that expected under reference conditions (Tables 1, S1).

2.4. Analysis

To compare the sensitivity scoring methods and assess regional effects, we focused on species with conservatism and model-based values in both regions (Boreal and Parkland). The same 123 native species were analyzed across the four combinations of scoring method and region (Tables 1, S1).

Model estimates were converted to the conservatism scale under the assumption that species increasing with human footprint are less conservative (less sensitive) and species decreasing with human footprint are more conservative (more sensitive). Therefore, increasing abundance ratios (current to reference) earned lower scores and decreasing ratios earned higher scores, which seemed to be a reasonable assumption (Fig. S1). We started by giving a 5 to the 1:1 ratio (90–110%) assuming that similar abundance under current and reference conditions suggests the species is neither tolerant nor sensitive. We then assigned integers in 20% increments above and below this middle range, so that with increasing current abundance relative to reference, species earned scores of 4 (110–130%), 3 (130–150%), 2 (150–170%), 1 (170–190%), or 0 (> 190%), and with decreasing current abundance relative to reference, species earned 6 (90–70%), 7 (70–50%), 8 (50–30%), 9 (30–10%), or 10 (< 10%).

We plotted frequency (number of species) distributions of the difference between conservatism and model-based scores by region and tested the rank-order correlation (Spearman test) between scores, interpreting significant positive correlation as general agreement. Because highly invasive or ruderal native species were ranked no lower than 1 by expert opinion, we ran the correlation tests before and after removing species that had a modelling score of zero. Additionally, we used Wilcoxon rank-sum tests to directly compare the distribution

Table 1
Expert opinion and model-based sensitivity scores for native wetland macrophytes in two regions of Alberta, Canada, showing the first 23 species in alphabetical order as examples (see Table S1 for further results). 'Expert' – median coefficient of conservatism from a max of nine botanists; 'Model' – ratio (%) of estimated abundance under current landscape conditions to that predicted under no human footprint, and the conversion to integer scores as explained in Section 2.4.

Species [a]	Boreal		Parkland	
	Expert	Model (%)	Expert	Model (%)
Achillea alpina	4	6 (89.2)	3	6 (70.3)
Agrostis scabra	2	5 (103.0)	1	4 (115.2)
Alnus incana	4	5 (108.4)	4	6 (80.7)
Alnus viridis	3	5 (94.8)	3	7 (60.1)
Alopecurus aequalis	4	6 (88.1)	4	5 (95.4)
Amelanchier alnifolia	4	5 (99.4)	2	5 (91.9)
Arctostaphylos uva-ursi	4	5 (106.7)	5	7 (51.8)
Artemisia biennis	2	5 (105.4)	2	5 (109.4)
Beckmannia syzigachne	3	5 (107.0)	2	4 (113.4)
Bidens cernua	3	5 (106.6)	3	3 (148.4)
Bolboschoenus maritimus	6	6 (71.0)	5	6 (87.1)
Calamagrostis canadensis	1	5 (101.9)	3	4 (111.8)
Calamagrostis stricta	4	5 (93.3)	3	6 (77.2)
Calla palustris	7	5 (95.3)	7	8 (46.8)
Caltha palustris	6	6 (88.1)	6	8 (35.8)
Carex aquatilis	2	5 (93.9)	3	7 (58.4)
Carex atherodes	5	4 (110.5)	5	3 (143.6)
Carex aurea	4	5 (103.2)	3	4 (116.7)
Carex bebbii	4	5 (106.6)	3	4 (114.4)
Carex brunnescens	6	5 (91.7)	5	9 (16.2)
Carex canescens	6	5 (91.2)	6	9 (26.3)
Carex diandra	5	5 (90.3)	5	9 (29.1)
Carex disperma	6	6 (88.9)	6	9 (21.4)

[a] Taxonomy follows the Flora of North America (FNA, 1993+), or the Integrated Taxonomic Information System (available at: http://www.itis.gov) for taxa that have not yet been treated in the Flora of North America, or for which the Flora treatment is out of date.

shapes of sensitivity values between regions under each scoring method.

3. Results

Distributions of sensitivity values (Fig. 2), positive rank-order correlations (Fig. 3), and abundance ratios negatively related to conservatism (Fig. S1) all indicated general agreement between the expert opinion and model-based methods. Correlation results (Boreal $r_s = 0.24$, $P = 0.009$; Parkland $r_s = 0.51$, $P < 0.001$) before removal of species (*Carex rostrata*, *Hieracium umbellatum*, *Phalaris arundinacea*, *Salix interior*) that had a modelling score of zero were similar to results after the removal (Fig. 3).

The level of agreement varied strongly by region. In the Boreal conservatism showed more spread than modelling scores (Fig. 2a–b), yet both methods had the same mode, and most (85%) differences were small (±2). Score distributions in the Parkland were similar for each method but shifted to lower values under expert opinion and higher values under modelling (Fig. 2d–e). Parkland also contained two large mismatches (Fig. 2f), including *Carex rostrata* that botanists ranked as relatively conservative (median of 8) despite it showing increased abundance under current conditions (239% or > 2× the reference abundance, Table S1), and *Epilobium palustre* ranked as tolerant by botanists (median of 1) despite a current abundance that was only 29% of the reference abundance (Table S1), the latter suggesting high sensitivity to environmental alteration.

Conservatism distributions for species were broadly similar between regions (Fig. 2a, d, Wilcoxon rank-sum test $P = 0.205$) whereas model-based score distributions differed between regions (Fig. 2b, e, Wilcoxon rank-sum test $P < 0.001$). The Parkland showed a stronger correlation (Fig. 3) and negative linear relationship (Fig. S1), but the Boreal had a smaller average difference in scoring results (Fig. 2c vs. f), so it was not immediately clear which region supported the better agreement.

4. Discussion

We compared expert opinion and modelling approaches for estimating species sensitivity to anthropogenic disturbance, focusing on wetland macrophytes at the prairie-boreal transition in western Canada. The scoring methods were rank-correlated with mode differences at or near zero, but there was also a strong regional effect (in the modelling) and large discrepancy for a few species. Overall the results indicate a general agreement between very different approaches to sensitivity estimation, suggesting they may be used interchangeably or complementarily in targeted monitoring for conservation (sensu Nichols and Williams, 2006; Bal et al., 2018).

4.1. Outlier cases

Out of 123 species, there were only two large discrepancies, both in the Parkland, where *Carex rostrata* received a high conservatism but low modelling score and *Epilobium palustre* received a low conservatism but high modelling score. Scores for *C. rostrata* may have been affected by taxonomic updates, limited detections, and changing perceptions of conservation status. *Carex rostrata* can easily be confused with the more common *C. utriculata*. These species are very similar in appearance and were formerly lumped as *C. rostrata*, and many regional botanical references refer only to *C. rostrata*. The relatively small number of *C. rostrata* records and large number of *C. utriculata* records in ABMI data suggests they were being distinguished, but it is possible that some *C. rostrata* records are misidentifications. Considering that this species is relatively uncommon in the Parkland region, even a small number of misidentifications could disproportionately affect modelling accuracy. Additionally, the provincial conservation status of *C. rostrata* recently shifted from Imperiled (S2) to Apparently Secure (S4) under NatureServe, and so taking the historical view (e.g. Vujnovic and Gould, 2002) could have led botanists to overvalue *C. rostrata* conservatism. Life history and ecological information on *C. rostrata*, such as rapid spread from rhizomes (Bernard, 1990) and moderate tolerance of grazing (Allen and Marlow, 1994), further suggests it was overvalued by botanists, but this information was published before the taxonomic split and likely applies to *C. utriculata* instead.

Table 2
Covariates used for modelling species sensitivity. Types of vegetation, soil, and human footprint were quantified by area in a 250 m buffer around the wetland polygon.

Physical-chemical	Water temperature; pH; salinity; dissolved oxygen and organic carbon; total nitrogen and phosphorus
Bioclimatic	Mean annual temperature and precipitation; mean warmest and coldest month temperatures; frost-free period; annual heat-moisture index; reference atmospheric evaporative demand
Surrounding vegetation[a]	Upland spruce and pine; lowland spruce and larch; forested peatlands; deciduous and mixedwood stands
Surrounding soils[b]	Productive; rapid drain; clay/saline
Human footprint	Total (alienating + successional); alienating (permanently altered vegetation); successional (post-land use regenerating vegetation)

[a] Vegetation types were considered more informative (i.e. able to discriminate) for modelling in the Boreal region and at some Parkland sites.
[b] Soil types were considered more informative for modelling at most Parkland sites.

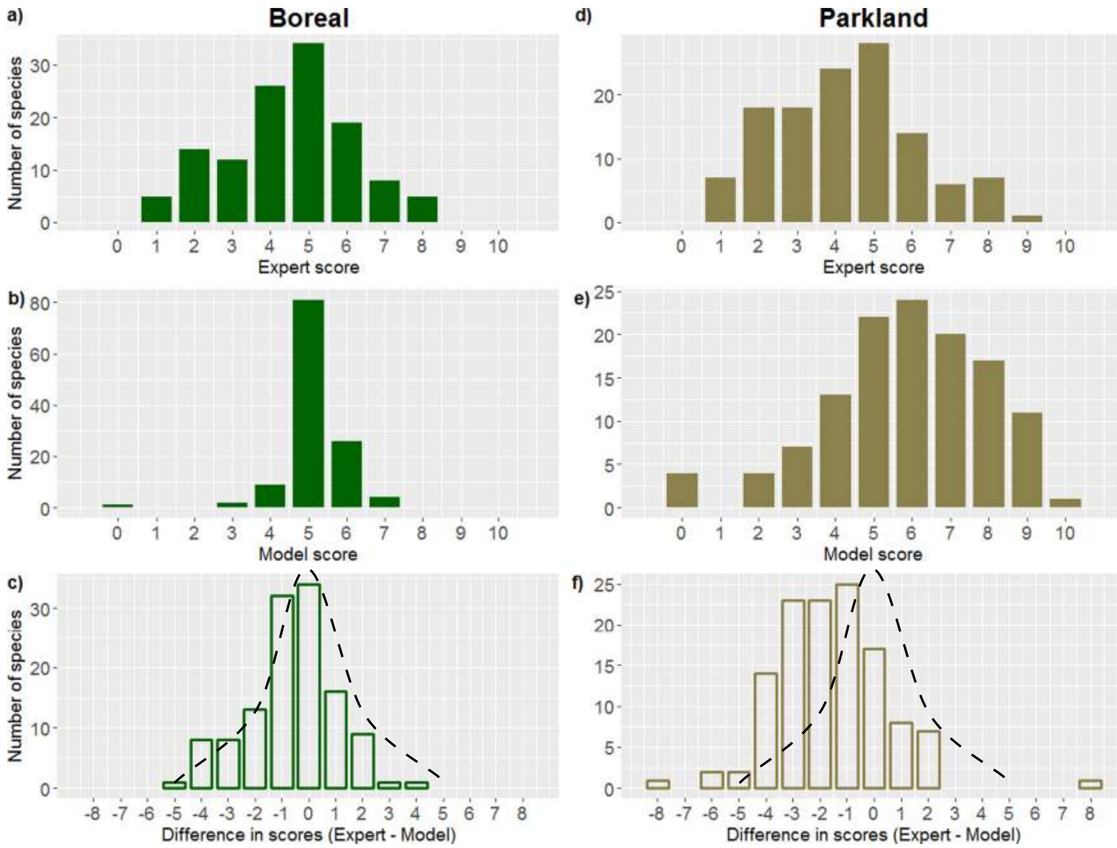

Fig. 2. Distributions of expert opinion (conservatism), model-based sensitivity, and their difference (Expert minus Model) for wetland macrophytes of the Boreal (a–c) and Parkland (d–f) regions of Alberta, Canada. The same set of 123 species was used for each plot. The bell curve illustrates a hypothetical distribution for good agreement.

From previous analyses of ABMI data, *Epilobium palustre* appears especially sensitive to ongoing conversion of grasslands to cultivation, which could explain its high model-based score in the agriculture-dominated Parkland. However, this species is not particularly common in the Parkland, which may have affected the accuracy of modelling and expert opinion. *E. palustre* was scored by only five of the nine botanists and over a wide range (1 to 8), indicating limited familiarity with the species and strong differences of opinion. Importantly, both cases (*Carex rostrata* and *Epilobium palustre*) are outliers and do not reflect patterns of bias or error, as in the systematic undervaluing of woody relative to herbaceous species in forests and wetlands across Illinois (Matthews et al., 2015).

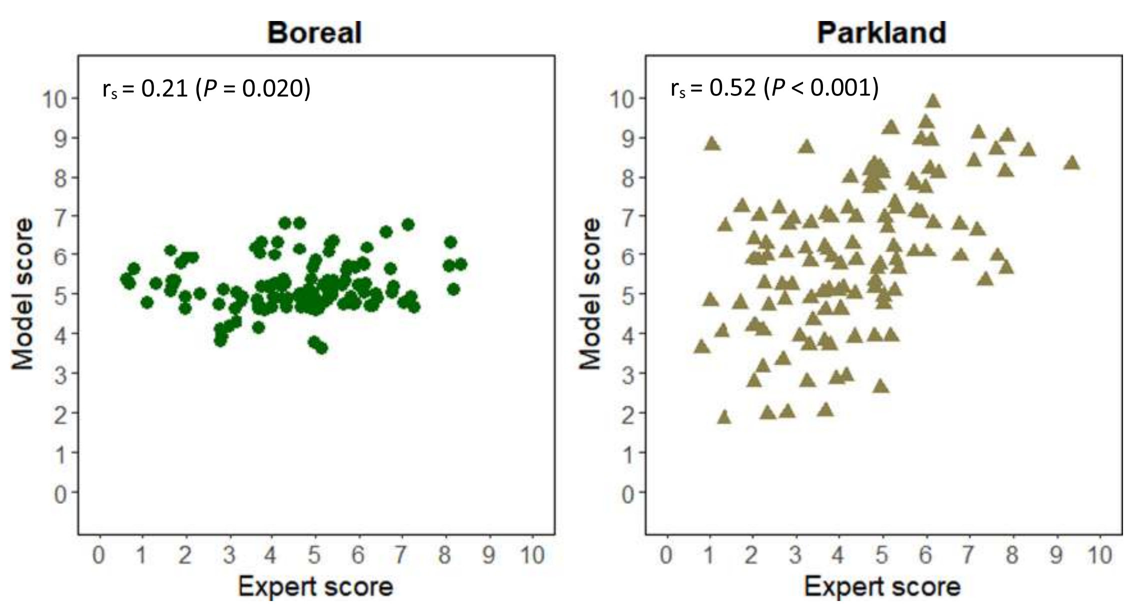

Fig. 3. Relationships between model and expert (conservatism) sensitivity values, with points jittered to show individual species positions. Species with zeros in modelling were removed because expert scores could go no lower than 1. Annotations indicate Spearman test results.

4.2. Limitations and conservatism validation

Both methods have their limitations and lacked measures of uncertainty. Although the species-environment relationship is modelled using local wetland data (i.e. water quality), quantification of current and reference conditions is based strictly on the surrounding landscape and not direct local observations. Given the oppositely skewed distributions in Fig. 2d and e, incorporating water quality information under current and reference conditions for the agriculture-dominated Parkland may have lowered the average modelling scores and increased agreement with the conservatism values. ABMI (2017c) and Nielsen et al. (2007) provide further discussion of the modelling limitations. The reliability of conservatism scores is obviously limited by how much familiarity botanists have with any given species, and botanists did not include confidence ranks (see Bried et al., 2012) with their scores. In addition, conservatism values are fixed across large heterogeneous areas with varying ecological opportunity (sensu Stroud and Losos, 2016), and niches are unlikely to be spatially constant at the species level (Leibold and Chase, 2017). Given the limitations with each method, and without measures of uncertainty, it is difficult to interpret if one method validates the other.

Nevertheless, because sensitivity modelling is objective and incorporates local and landscape environmental variation (i.e. allows varying ecological opportunity), some might view this method as providing a validation test for subjective regional conservatism values. Many wetland studies have validated the average conservatism of species assemblages using a "dose-response" analysis, positioning sites along a gradient of environmental degradation (e.g. Lopez and Fennessy, 2002; Bourdaghs et al., 2006; Ervin et al., 2006; Miller and Wardrop, 2006; Bried et al., 2013; Wilson et al., 2013; but see Bried et al., 2016; Jog et al., 2017). Individual conservatism scores have also been validated using the dose-response gradient approach (Mushet et al., 2002; Cohen et al., 2004; Bowers and Boutin, 2008), along with null modelling of species co-occurrence (Matthews et al., 2015) and trait measurements associated with key life history trade-offs (Bauer et al., 2017). Together with these studies, we conclude that individual coefficients of conservatism, despite their subjectivity, carry considerable ecological information and correspond with anthropogenic impacts to wetlands, supporting average conservatism as an indicator for site-specific and large-scale wetland biomonitoring (DeBerry et al., 2015; USEPA, 2016).

4.3. Regional significance

Regional stratification helps reduce environmental noise in freshwater biodiversity assessments (Stoddard, 2005), and Alberta's Natural Regions (including Boreal, Parkland, and several others) and Sub-regions provide environmental context and stratification for resource planning and management throughout the province (NRC, 2006; McNeil, 2008). Our analysis found clear regional differences in model-based sensitivity but not in conservatism opinions. The modelling incorporated detailed landscape geospatial data and local field-collected data from dozens (Parkland) or hundreds (Boreal) of wetlands, capturing heterogeneous environmental conditions within each region. Based on the modelling there was a much narrower distribution of sensitivity in the Boreal than in the Parkland, possibly reflecting a more limited range of environmental conditions shaping regional species distributions. The large proportion of intermediate values in Boreal modelling (Fig. 2b) indicated similar current and reference abundances, suggesting moderate sensitivity for many species or simply reflecting a minimum human footprint. In contrast the Parkland region has a proportionately large human footprint (ABMI, 2017a), which may have facilitated the wider range of model output and greater number of low-valued species compared to the Boreal. By capturing regional differences in response variation, sensitivity modelling seems well suited for regional conservation planning and tailoring species-specific actions to different regions.

In contrast to modelling, the conservatism values were assigned as static within regions, implicitly assuming species vary only according to anthropogenic factors and not the natural heterogeneity present at local and landscape scales ("within-region neutrality"). Just as conservatism is unlikely to hold across regions or broad latitudes due to climatic and environmental heterogeneity (Nichols, 1999; Johnston et al., 2010; Spyreas, 2014; but see Chamberlain and Ingram, 2012), within-region neutrality also seems untenable given strong local and landscape variation. Still, it is commendable that botanists attempted regional-based scores in this case (Forrest, 2010), as most conservatism assignments have occurred throughout large geopolitical boundaries and not with respect to ecologically meaningful regions or management units (Spyreas, 2014).

5. Conclusions

Expert field botanists cannot possibly consider all the modifying variables and subtle environmental heterogeneity captured in modelling. Instead botanists use their cumulative field experience, sometimes combined with knowledge of plant life history and ecology, to interpret the average conditions under which they encounter each species. Likewise, modelling is only as good as the input data and knowledge of the study system. There will always be cases in which neither method is accurate, such as rare or elusive species where experts lack the experience to make an informed decision and empirical results become imprecise (Nielsen et al., 2007; Matthews et al., 2015). However, it is patterns of excessive mismatch that signal deficiencies in expert knowledge, errors in the modelling process, or both. Moreover, the species sensitivity values are often aggregated in practice (DeBerry et al., 2015; ABMI, 2016; USEPA, 2016), diminishing the importance of individual species errors.

Whether species are overvalued or undervalued may come down to one's trust in experts vs. models. Comparing Fig. 2d and e, for example, believers of informed opinion might conclude that modelling tended to overvalue, whereas adherents of modelling might conclude that botanists tended to undervalue. Fortunately, the preference becomes inconsequential when both are reliable and can avoid systematic disagreement. Only when agreement is weak does it seem worth asking which approach is better or applying both to help offset their disadvantages.

Monitoring how species respond (e.g. positively or negatively, linearly or non-linearly) to environmental degradation and management interventions is germane to triggering conservation actions and adjusting management practices (Nichols and Williams, 2006; Westgate et al., 2013). Both sensitivity measures used here could work well for bioassessments, especially combined with cutting-edge ideas and analyses for taxonomic surrogacy and management prioritization (Tulloch et al., 2013; Lindenmayer et al., 2015; Westgate et al., 2017; Bal et al., 2018). Conservatism-based floristic quality assessments move beyond traditional measures of biodiversity change (e.g. richness; Hillebrand et al., 2018), but because conservatism values are fixed over large areas and may perform similarly with and without abundance-weighting (DeBerry et al., 2015; Kutcher and Forrester, 2018), detecting changes in local biological condition generally requires a shift in species composition. In contrast, sensitivity modelling incorporates detailed information for assessing impacts spatially and over time, as seen in the ABMI program. A complementary approach of expert opinion and modelling may therefore provide the ideal framework for conservation assessments based on species sensitivity.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2018.07.013.

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