Environmental Reviews

Quantifying the impacts of oil sands development on wildlife: perspectives from impact assessments.

Journal:	<i>Environmental Reviews</i>
Manuscript ID	er-2018-0118.R1
Manuscript Type:	Review
Date Submitted by the Author:	29-Jan-2019
Complete List of Authors:	Campbell, Mac; University of British Columbia Kopach, Brian; MSES Inc. Komers, Petr; MSES Inc. Ford, Adam; University of British Columbia, Department of Biology
Is this manuscript invited for consideration in a Special Issue?:	Not applicable (regular submission)
Keyword:	boreal forest, habitat suitability index, industrial impact, mitigation, resource selection function

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Title: Quantifying the impacts of oil sands development on wildlife: perspectives from impact assessments.

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Abstract

Anthropogenic landscape disturbances, including industrial development, can have significant impacts on wildlife populations. In Canada, federal, territorial, and provincial laws require major industrial development projects to submit detailed environmental impact assessments (EIA) reports as part of the project application process. These assessments are meant to establish baseline habitat conditions and predict which landscape components will be altered by the project and to what degree. Based on these changes, indirect predictions for wildlife impacts are made using a variety of models, which can vary in validation adequacy and often rely heavily on expert opinion. In the oil sands region of Canada, wildlife species and habitat types used to make predictions are not comprehensive nor standardized between EIAs, despite a high degree of landscape similarity between projects. We extracted habitat model parameters, projected impacts, and anticipated mitigation effectiveness from 30 project EIAs. Despite all these projects occurring in the same natural region, we found very little agreement in the species used to assess wildlife impacts as well as the parameters used to model impacts on those species. Relative to unvalidated habitat models, we found that models receiving independent validation required half the habitat amount for proponents to conclude that the project will have an adverse effect. Our analyses have exposed a number of areas where policy could improve the efficiency of the EIA process as well as the scientific rigour underlying regulatory decisions.

Keywords: boreal forest, habitat suitability index, industrial impact, mitigation, resource selection function, validation

Introduction

Loss of habitat is driving the decline of populations and the extinction of species at rates well exceeding the background extinction rate measured in the fossil record (Barnosky et al. 2011; Ceballos et al. 2017). The cumulative impacts of anthropogenic disturbances are associated with range contractions (Shackelford et al. 2018) and other indicators of population declines (Lamb et al. 2017). Human-caused disturbances reduce the abundance of wildlife (Stewart and Komers 2017), the extent to which animals move through the landscape (Tucker et al. 2018), and is causing species to shift their activity times to the night when people are less active (Gaynor et al. 2018). Human-caused disturbances also alter ecological interactions, such as predator-prey dynamics (Wilman and Wilman 2017; Fisher and Burton 2018). Overall, human impacts on wildlife have the potential to cause negative ecological and socioeconomic impacts (Ceballos et al. 2017).

To counteract habitat loss and other negative impacts on wildlife, a number of international, multilateral agreements have been developed to target minimal proportions of the landscape that must be protected from threats, including industrial development. For example, the Convention on Biological Diversity (CBD) Strategic Plan for Biodiversity 2011-2020 (Convention on Biological Diversity 2010) recommends a minimum of 17% of land be protected by each country. Signatories to the CBD, such as Canada, are working to identify areas where the creation of new protected areas will have a positive impact on biodiversity (Coristine et al. 2018). Outside of protected areas, habitat protection has been highly effective through management by Indigenous people (Garnett et al. 2018). However, in areas where industrial-scale resource extraction is a dominant form of land use, habitat and wildlife protections are more uncertain.

In Canada, protection of biodiversity outside of protected areas is embedded in the regulation of large-scale industrial projects through the Environmental Impact Assessment (EIA) process. The EIA process in Canada includes plans for mitigation that begin with the establishment of baseline (i.e., pre-construction) environmental conditions, then a prediction of how the construction and operations-related phases of the project will change baseline conditions, followed by the predicted effectiveness of mitigation measures.

One of the focal areas of most EIAs is wildlife, as reflected in guidelines for preparing EIAs (Canadian Environmental Assessment Agency 2008): “A description of the physical and biological setting, including the physical and biological components in the area that may be adversely affected by the project (e.g., air, fish, terrain, vegetation, water, *wildlife*, including migratory birds, and *known habitat use*).” [emphasis added]. The EIA process is targeted at a subset of species or taxa that occur in a project area, usually referred to as valued ecosystem components (VECs). VECs vary by project, and may include harvested, culturally-important species, and/or species at risk to reduce exposure to liability (Beanlands and Duinker 1983; Ball et al. 2013; Murray et al. 2018). VECs are selected by the proponent, though consultation with Indigenous communities and stakeholders (i.e., hunting groups, members of the general public, governments, industry representatives, lawyers, and scientists) is often part of the selection process (Bangert and Slobodchikoff 2004; Murray et al. 2018). Participation in VEC selection varies, with proponents exercising discretion over which communities are consulted and the degree to which substantive consultation is actually pursued (Fitzpatrick and Sinclair 2009; Bond et al. 2014). There is general lack of clear direction or guidelines to select VECs, which has undermined the rigour of environmental assessment work (Olagunju and Gunn 2015).

The impact of a project on VECs is rarely assessed for population vital rates. For example, the effect of the project on a change in population density or vital rates is not provided. Instead, impacts on wildlife are measured indirectly through predicted changes in habitat amount. Typically, habitat assessments are derived from habitat suitability indices (HSI) and resource selection functions (RSF) (Muir et al. 2011). HSI models can be (and often are) based on expert opinion, which may be then validated with an independent dataset (U.S. Fish and Wildlife Service 1981; Muir et al. 2011). HSIs are commonly criticized for subjectivity of assigning suitability scores to habitat types and their inadequate validation of assumptions using independent data (Burgman et al. 2001; Muir et al. 2011). In contrast to HSI models, RSF models use empirical data to predict habitat selection based on wildlife presence/absence or use/availability of landscape features (Muir et al. 2011). Empirical data for these models are collected using radio or satellite telemetry, camera traps, and indices of relative density (Muir et al. 2012). The accuracy of habitat models is a critical requirement for understanding the location and amount of habitat loss and the types of mitigation required to minimize negative impacts on wildlife (Barker and Jones 2013; Murray et al. 2018; Fisher and Burton 2018; Hrudehy et al. 2010).

Currently, federal laws governing environmental assessments in Canada are undergoing review, with calls to make this process better informed by science (Jacob et al. 2018; Westwood et al. 2019). Moreover, Canada has the highest global contribution to wilderness of any country, yet many of these areas are vulnerable to impacts from resource extraction (Venter et al. 2016; Lamb et al. 2018). It is therefore important to consider how impacts to wildlife are assessed in large-scale industrial projects, especially in remote landscapes like northern Alberta, Canada where development of oil sands conflicts with the recovery of species like woodland caribou

(Hebblewhite 2017). Northern Alberta has one of the fastest rates of land-cover disturbance on the globe (Komers and Stanojevic 2013), which is likely to continue in the future. The Alberta Energy Regulator (AER), which oversees development in this region, currently lists 9 approved surface mines, over 50 approved thermal in-situ projects, and over 200 primary and enhanced bitumen recovery projects (Alberta Energy Regulator 2018a). The AER states that approximately 24% (90,000 km²) of Alberta's boreal forest contains productive oil sands deposits (Alberta Energy Regulator 2018b). As such, northern Alberta is one of the most active regions of environmental regulation and resource extraction in North America. Unfortunately, existing land use plans (i.e. Lower Athabasca Regional Plan) provide no guidance on how wildlife effects should be assessed, and data from regional monitoring programs (i.e Alberta Biodiversity Monitoring Institute) operate at scales too large for the assessment of project specific effects. The lack of regional frameworks or suitable data are factors underlying inconsistency in the way project-related impacts are assessed for oil sands developments in Alberta.

Here, we examined the assessment of wildlife impacts reported in EIAs developed for oil sands projects in the boreal forest of northern Alberta. While these EIAs may vary slightly in their project-specific Terms of Reference and geographic scope, this region contains reasonably similar landscapes, flora, and fauna. For this reason, we hypothesized that EIAs should adopt similar methods and approaches under the regulatory principle that promotes use of the 'best available evidence' (Government of Canada 2018; Westwood et al. 2019). Specifically, we anticipated that EIAs would use similar VECs, habitat types, and suitability values assigned to those habitat types, would validate habitat models, and would accommodate uncertainty in the prediction of how effective mitigation would be based on habitat models. Habitat modelling approaches used by EIAs underlies the interpretation of baseline conditions, impacts, and

effectiveness of mitigation measures aimed at protecting wildlife. As such, habitat models are central to understanding how science shapes Canada’s EIA process.

Approach to the review

We examined 30 of the most recent and publically-available EIAs that were submitted and posted to the government registry in 2004-2017. These EIAs were selected from the Alberta Government’s list of Completed EIAs by date and accessed either online or through Management and Solutions in Environmental Science (MSES Inc.), who performed a number of third-party reviews on EIAs. The 30 projects were located within an ~11,000,000 ha area of the Boreal Forest Natural Region of Alberta (Fig. 1). Within the Boreal Forest Natural Region, there are four sub-regions represented by the EIAs: Central Mixedwood (23 projects), Dry Mixedwood (2 projects), Lower Boreal Highland (4 projects), and Upper Boreal Highland (1 project). All sub-regions contain slight variations in their proportions of aspen, balsam poplar, white spruce, black spruce, and jack pine forests, with more lodgepole pine stands in the highland sub-regions. Wetlands and other water bodies are quite common in all sub-regions, with wetlands dominating 15 to >50% of areas within our dataset (Alberta 2015). Thus, while there is some diversity of habitat types represented among these EIAs, all occur in the same natural region and most (76%) occur in the same natural sub-region.

For each project, we recorded the year of submission, latitude and longitude, project footprint (ha), local study area [LSA] (ha), regional study area [RSA] (ha), years to project completion and output (Barrels per day) (Supplementary Table S1). Although assessments are usually done at both the LSA and RSA scales, we only analysed habitat changes at the LSA scale. This scale enables us to explore the projected impacts of the proposed project, rather than

161 projects within the regional study area. We recorded the VECs listed in the wildlife section of
162 each EIA. For each VEC, we recorded the total predicted habitat loss (ha), along with predicted
163 losses for each quality component of the total (i.e., high, moderate-high, moderate, moderate-low
164 and low quality habitats). The evaluations for effect magnitude (either negligible, low-negligible,
165 low, moderate-low, moderate, moderate-high or high) and predicted residual effects (significant
166 or not significant) were also recorded for each VEC. These evaluations were made within the
167 EIA by the proponent and were based on predicted habitat losses (Supplementary Table S2).

168 We extracted the method used (HSI, RSF, RSPF, occupancy analysis) to assess habitat
169 amount and loss (Supplementary Table S3). For each habitat model, we extracted the habitat type
170 (e.g., vegetation class, distance to water) and the coefficient used for each habitat type (i.e., the
171 ‘betas’ in the RSF or the suitability score for HSIs). In some cases, these data were not explicitly
172 provided within the EIA, but rather, referenced literature not included in the EIA. We searched
173 these external documents and where possible, recorded the habitat type and coefficient. In some
174 cases, it was unclear which habitat model was selected from the external literature for use in the
175 EIA. For example, an EIA may have referred to a separate study that had more than one habitat
176 value listed for a given species (e.g., due to sex, or season-specific habitat models), but did not
177 define which value was chosen for the EIA. We noted in our analysis when a habitat type was
178 referred to but the values were not shown or available. We only considered a model to be
179 validated if it was tested with independent field data (Muir et al. 2011).

180 We tested if validation – an indicator of scientific rigour – predicted the conclusion of the
181 EIA for each VEC. We created two generalized linear models, both using the amount of habitat
182 lost and validation (yes/no) as predictor variables. For one model, we used the predicted impact
183 of the project on the VEC during the construction and operation phases (i.e., the project impact)

as a binary response variable (i.e., high vs moderate/low impact). For the second model, we used the predicted impact of the project on the VEC following mitigation (i.e., the residual effects) as a binary response variable (i.e., significant or not significant). We used a generalized linear model with a binomial distribution and logit link to quantify the effects of habitat loss and validation on these respective outcomes. We used the statistical software program R (R Core Team 2018) for this analysis.

Findings of the review

The combined footprint area from the proposed projects represented by the 30 EIAs was 99,195 ha, with a total combined LSA of 728,291 ha (Table S1). The projects had a mean operational lifespan of 34.1 years (SD = 7.8), with a mean output of 124,996 barrels per day (SD = 84,429). We identified 35 wildlife VECs, which included 30 single species (e.g., moose) and 5 aggregate taxa (e.g., old growth forest bird community). There were 6 - 15 VECs per project (Fig. S1), with a mean of 10.23 (SD = 2.14). Conversely, the mean number of projects using a given VEC was 8.77 (SD = 8.09) over a range of 1 to 30 projects (Fig. 2). Moose was the only VEC used in all EIAs and only 10 VECs were chosen by $\geq 50\%$ of the EIAs (Fig. 2).

The total predicted habitat loss for the 35 VECs was 881,208 ha, 51% of which was from the high quality category (Fig. 3). The 'Moderate-high' and 'Moderate-low' habitat quality categories were only used in $\sim 30\%$ of the 316 VEC habitat loss evaluations, which explains their relatively low values in Fig. 3. The mean predicted habitat loss was 3048 (SD = 2244) ha per VEC per project (Fig. 4a), while the mean predicted high quality habitat loss was 1643 (SD = 1709) ha per VEC per project (Fig. 4b). Although all predictions are reported here in hectares, 4 projects (Carmon Creek SAGD, Kai Kos Dehseh SAGD, Joslyn North Mine and Kearl) used

habitat units (HU) instead, rather than an areal unit. A HU was defined as $\Sigma (HSI_i \times A_i)$, where HSI_i = the habitat suitability index for habitat type polygon i and A_i = the area of that polygon (Imperial Oil 2005).

Impacts to wildlife habitat were estimated from HSI models for ~94% of VEC assessments. RSF and resource selection probability function (RSPF) models were used in 3% and 2% of predictions, respectively. Of the 316 VEC habitat models described, 12% ($n = 39$) were validated. There were 1664 unique habitat types (e.g., “blueberry aspen-white spruce”) described among all models, of which 58% were not used by more than one EIA. The number of habitat types used per project (aggregated among all VECs in a project) varied from a maximum of 262 to a minimum of 44, with a mean of 116.3 ($SD = 63.0$).

A total of 6227 VEC~habitat type relationships were described by the EIAs (e.g., “black bear ~blueberry aspen-white spruce”), of which 79% were not used by more than one EIA. Individual VEC~habitat type relationships were used in a minimum of 1 to a maximum of 13 projects, with a mean occurrence in 1.4 ($SD = 1.1$) projects.

At the finest scale of specificity, we found 6816 unique VEC~habitat type~coefficient (e.g., “black bear ~blueberry aspen-white spruce~0.73”), of which 30% had no reported value in the project EIA itself or material referenced by the project EIA. Of these 6816 unique VEC~habitat type~habitat values, 82% were used by no more than one EIA.

During project construction and operation phases of the project, 19% ($n=59$) of VECs were predicted to experience high negative impact and 39% ($n=123$) were predicted to experience moderately-negative impacts. Following mitigation and reclamation efforts, 10% ($n=33$) of VECs were predicted to experience significant residual effects. In other words, mitigation efforts are predicted to successfully reverse the highly-negative and moderately-

negative impacts for 72% of VECs. Moreover, 29 of 30 EIAs predicted a high or moderate impact during construction and operations for at least one VEC, but only 7 EIAs predicted significant residual effects (post-mitigation) for at least one VEC.

The amount of predicted habitat loss had a positive effect on the probability that the EIA concluded significant negative ($\beta = -5.892\text{e-}05$; $P = 0.006$) impacts (i.e., during construction and monitoring) and significant negative ($\beta = -6.445\text{e-}05$; $P = 0.004$) residual effects (i.e., post-mitigation); however, this relationship was stronger if a model was validated (Fig. 5). If a VEC model was validated, the probability that the EIA concluded either significant impacts (Fig. 5a) or significant residual effects (Fig. 5b) was about twice as great for a given amount of predicted habitat loss. In other words, to be considered significant, habitat loss for an unvalidated model had to be 100% greater when compared to a validated model.

Discussion

Our results indicate pervasive subjectivity, lack of consistency, and specious rigour in estimating the impacts of large resource extraction projects on wildlife in Canada. In spite of similar projects occurring over similar timeframes, and in similar landscapes, the tendency was for EIAs to assess project specific: 1) number and type of VECs; 2) habitat types and coefficients selected to quantify species-habitat relationships; 3) habitat modelling methods; 4) choice of validation procedures, if any, to assess the performance of habitat models. Associated with the high degree of variability in model design, EIAs have a high degree of confidence in the effectiveness of mitigation measures even though there are few empirical examples demonstrating successful restoration outcomes in this sector (Vitt and House 2015; Finnegan et al. 2018). Our results suggest that EIA reform in Canada requires a significant infusion of

science if evidence-based decision making – a stated goal of policy (Government of Canada 2018; Jacob et al. 2018; Westwood et al. 2019) - is going to be realized.

We were surprised to find that the subjective habitat models did not share more agreement between EIAs. Expert opinion based HSI models were used for 94% of VEC models, yet only 17% of these HSI models were the same in more than one project. We expected that empirical models of habitat selection to show greater disagreement in predictor variables because of variation in field collection and statistical methods. However, expert opinion HSI models are inherently subjective (Johnson and Gillingham 2004), so we would expect that their authors would adopt the methods of previous EIAs or peer-reviewed habitat models. It is not clear why there is such little agreement in how to create habitat models for EIAs in the oil-sands generally, and for the 35 VECs we identified specifically. Since all 30 EIAs were completed for projects existing within the same Boreal Forest Natural Region, it is unlikely that ecosystem variation alone caused the considerable discrepancies we found. One explanation is that there is a lack of practitioner expertise or coordination, though this seems unlikely. Many of the practitioners who create EIAs are affiliated with professional societies that have competency, training, and professional development standards that are required for membership. Lack of awareness and coordination has been cited as one reason for such broadscale inconsistencies in EIA practices in Canada (Lott and Jones 2010; Olagunju and Gunn 2015). With such a high degree of subjectivity in methods, impact predictions likely underestimate the significance of wildlife habitat loss, and predictions for mitigation or reclamation may be insufficient.

Subjectivity in the HSI-based models is not inherently flawed. All habitat models – empirical or opinion based – are estimates of how organisms interact with spatial variation in environmental conditions. However, confidence in models depends on the degree to which the

are validated and an assessment of model performance is provided. For empirical habitat models, k-fold cross validation, or other statistical methods (e.g., Area Under the Curve) are used to assess performance (Johnson and Gillingham 2004; Wiens et al. 2008). For HSIs, independent data, such as camera traps, have been used to test, for example, if relative activity increases positively with the HSI score (e.g., Table S1: Syncrude Mildred Lake Extension Project). It is unknown how the unvalidated models we reviewed would stand up to a test of validation, had this option been exercised. Given the largely inconsistent approaches used to measure and rank ‘habitat’, we have no basis with which to measure the performance, accuracy, or reliability of most habitat models used in oil sands EIAs.

Validation of habitat models appears to be associated with a more precautionous assessment of project impacts by EIA authors. EIAs generally follow a rubric to assign severity of impact or significance of residual effects. Total amount of habitat loss is, unsurprisingly, a strong predictor of the proponent reaching a conclusion that a project will have an adverse effect on wildlife. However, validation is associated with a 50% lower amount of habitat loss before such a conclusion is reached. This suggests that EIAs adopting more rigorous methods are also less confident in the effectiveness of mitigation. This link makes sense, as many proposed mitigation measures are not predicted to work until several decades after the operation phase of a project, meaning that it is 50-150 years before such results could possibly be measured (Rooney et al. 2012; Audet et al. 2015; Ketcheson et al. 2016; Rodríguez-Estival and Smits 2016).

While EIAs are more likely to find a significant negative effect on wildlife if the habitat model was validated, of the 1681 oil sands applications made to the AER since December 2013, 91% were approved, 1% were denied, and ~8% were closed or withdrawn (Alberta Energy Regulator, 2018c). In spite of the links between model validation, area of habitat lost, and

severity of impact, there is no obvious relationship between the severity of impact and project approval. In a recent review, (Orenstein (2018) found that 74% of energy projects entering the Canadian EIA process in 2012-2018, despite them reporting significant adverse affects. From a regulatory perspective, these projects were approved because the Governor-in-council determined that the adverse effects were justified for the greater good of society. This ‘greater good’ may not include people most directly impacted by the project, including local Indigenous communities (Gibson et al. 2016). Thus, it is not clear if or how reporting negative impacts on wildlife in an EIA has any bearing on project approval.

Our results have exposed multiple weaknesses in the Alberta oil sands wildlife EIA process. Improving on these weaknesses is necessary for a two key reasons. Firstly, habitat models inform predictions regarding the magnitude and duration of impacts on wildlife (Muir et al. 2011). Accurate prediction making is required to ensure the VECs are conserved, not only for ecological reasons, but also for reasons presented by various Indigenous communities and stakeholders during the public consultation period (e.g., cultural significance or harvesting value) (Pope et al. 2013; Nguyen et al. 2016; Environment and Climate Change Canada 2018). Indigenous communities and stakeholders must have confidence that mitigation measures will be successful, as they may be directly affected by any wildlife declines and/or extirpations that occur due to mismanagement (Moore et al. 2015). Secondly, the divergence in content among wildlife habitat models in EIAs represents cost inefficiencies for industry. Rather than paying multiple practitioners to construct bespoke habitat models, companies could rely on pools of existing data and scientific consensus regarding wildlife populations in the area (Retief et al. 2016). Such an approach would require industry coordination and some degree of collaboration, which may require leadership and support from a regulatory agency. Under the current process,

costs are incurred to proponents via project delays or denials due to insufficient EIA modelling and supplemental information requests by intervenors (Hrudey et al. 2010; Sánchez and Mitchell 2017).

Future improvements to the EIA process should include standardization of habitat models at the sub-region level and all models should be require empirical-validation. Such standardization would better reflect the landscape similarity between projects, would confer greater scientific credibility, improve conservation efficacy (Muir et al. 2012; Olagunju and Gunn 2015), and better satisfy cross-sectoral demand for increased rigour (Jacob et al. 2018; Westwood et al. 2019). Moving forward, our results suggest that a more open and substantive dialogue is needed between project proponents, Indigenous communities, scientists, and other stakeholders to meet domestic and international sustainability targets.

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Figures

Figure 1. Map showing the location of the 30 projects used in this review, with the Alberta natural subregions.

Figure 2. The number of taxa chosen for assessment in Environmental Impact Assessments in northern Alberta varied, shown as a) by the number of taxa selected per project and b) the projects per taxa.

Figure 3. Cumulative loss of habitat, by habitat quality, for all taxa combined, as predicted by Environmental Impact Assessments conducted in northern Alberta.

Figure 4. Boxplot showing the predicted loss of habitat per project for each species for (a) all habitat qualities and for (b) high-quality habitat only.

Figure 5. The probability of predicting a) a high impact during project construction and operation and b) a significant residual effect post mitigation and reclamation based on habitat loss between predictions supported by either validated (grey) or unvalidated (black) models.

Figure 1. Map showing the location of the 30 projects used in this review, with the Alberta natural subregions.

193x259mm (300 x 300 DPI)

Figure 2. The number of taxa chosen for assessment in Environmental Impact Assessments in northern Alberta varied, shown as a) by the number of taxa selected per project and b) the projects per taxa.

190x254mm (96 x 96 DPI)

Figure 3. Cumulative loss of habitat, by habitat quality, for all taxa combined, as predicted by Environmental Impact Assessments conducted in northern Alberta.

73x40mm (300 x 300 DPI)

Figure 4. Boxplot showing the predicted loss of habitat per project for each species for (a) all habitat qualities and for (b) high-quality habitat only.

190x254mm (96 x 96 DPI)

(a)

(b)

Figure 5. The probability of predicting a) a high impact during project construction and operation and b) a significant residual effect post mitigation and reclamation based on habitat loss between predictions supported by either validated (grey) or unvalidated (black) models.

190x254mm (96 x 96 DPI)