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REVIEW

Guidance for analytical methods to cumulative effects assessment for terrestrial species

C. Lisa Mahon and Shawna Pelech

Abstract: Landscapes in Canada are undergoing change due to resource and land use stressors and climate stressors. Understanding the cumulative effects of these stressors is challenging because of the complexity of ecosystems, the variability of stressors, and species response to individual or multiple stressors. A key challenge within the field of cumulative effects assessment (CEA) is guidance that describes and evaluates analytical methods. In this review we discuss four broad categories of methods with current or potential use for project-based and effects-based CEA for species in terrestrial systems: (*i*) qualitative review, (*ii*) habitat supply models, (*iii*) empirical species–stressor models, and (*iv*) decision support models. We describe each method and provide examples, highlight advantages and limitations, identify how methods address key science-based CEA questions, and provide direction on when and why to use specific CEA methods. Empirical species–stressor models and decision support models are the only analytical approaches that provide answers to many science-based CEA questions including how multiple stressors combine to affect an individual species and the certainty of multiple stressor effects. We provide recommendations for using one or more methods as complementary building blocks to fill data gaps, improve understanding and communication, engage diverse partner groups, and increase the quality and credibility of the CEA. Our review supports a move toward regional scale, effects-based CEA where partner collaboration to design, implement, and analyze comprehensive assessments of multiple stressors will (*i*) expand our knowledge of terrestrial species response to stressors and (*ii*) inform best management practices for resource industries and conservation and management actions for land managers.

Key words: analytical methods, Canada, cumulative effects assessment, models, stressors, terrestrial species.

Résumé : Les paysages au Canada sont en train de changer en raison de facteurs de stress liés aux ressources et à l'utilisation des terres et de facteurs de stress climatiques. Il est difficile de comprendre les effets cumulatifs de ces facteurs de stress en raison de la complexité des écosystèmes, de la variabilité des facteurs de stress et de la réponse des espèces à des facteurs de stress individuels ou multiples. Un des principaux défis dans le domaine de l'évaluation des effets cumulatifs (EEC) est l'orientation qui décrit et évalue les méthodes analytiques. Dans cette synthèse, les auteurs examinent quatre grandes catégories de méthodes actuellement ou potentiellement utilisées pour l'EEC des projets et des effets sur les espèces dans les systèmes terrestres : (*i*) l'examen qualitatif, (*ii*) les modèles d'approvisionnement de l'habitat, (*iii*) les modèles empiriques de facteurs de stress des espèces et (*iv*) les modèles d'aide à la décision. Ils décrivent chaque méthode et fournissent des exemples, soulignent les avantages et les limites, identifient comment les méthodes répondent aux questions clés de l'EEC basée sur la science, et fournissent une orientation sur le moment et la raison d'utiliser des méthodes d'ECC spécifiques. Les modèles empiriques de facteurs de stress des espèces et les modèles d'aide à la décision sont les seules approches analytiques qui fournissent des réponses à de nombreuses questions d'EEC fondées sur la science, notamment la manière dont de multiples facteurs de stress se combinent pour affecter une espèce individuelle et la certitude des effets de multiples facteurs de stress. Ils fournissent des recommandations quant à l'utilisation d'une ou de plusieurs méthodes comme éléments de base complémentaires pour combler les lacunes des données, améliorer la compréhension et la communication, engager divers groupes de partenaires et accroître la qualité et la crédibilité de l'EEC. Leur synthèse soutient un changement vers une EEC à l'échelle régionale, où la collaboration des partenaires pour concevoir, mettre en œuvre et analyser des évaluations complètes de multiples facteurs de stress permettra (*i*) d'élargir nos connaissances sur la réponse des espèces terrestres aux facteurs de stress et (*ii*) de définir les pratiques optimales de gestion pour les industries primaires et les mesures de conservation et de gestion pour les gestionnaires des terres. [Traduit par la Rédaction]

Mots-clés : méthodes analytiques, Canada, évaluation des effets cumulatifs, modèles, facteurs de stress, espèces terrestres.

1. Introduction

Landscapes throughout much of Canada have been subject to an increasing amount and variety of human-induced disturbances over the past decades. In many cases, these changes have resulted from a combination of resource sectors (forestry, oil/gas industries, mining, ranching) and land use practices (agriculture, urban and rural development) and their associated stressors (e.g., harvest units, agriculture plots, well sites, pipelines, industrial sites, mine sites, roads, powerlines, rail lines). In addition, climate change stressors or variables (e.g., temperature change,

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precipitation change, permafrost melt) and climate-induced changes to natural disturbance regimes (e.g., wildfire, floods, insect outbreaks, disease outbreaks) resulting from human-caused climate change are also influencing landscapes and habitats (Stralberg et al. 2018). Climate stressors may combine with existing resource and land use stressors to create novel and complex additive or interactive effects (Didham et al. 2007; Darling and Côté 2008). These individual and combined stressors alter the quantity and quality of natural habitats. Understanding the cumulative effects of stressors is challenging because of the complexity of ecosystems, the variability of stressors, and species response to individual or multiple stressors (Hodgson et al. 2019; Mahon et al. 2019).

Evaluating the combined or cumulative effects of these human-induced disturbances (hereafter stressors) on terrestrial species is now central to many conservation and management (Gunn et al. 2014), research and monitoring (Burton et al. 2014; Mahon et al. 2019), and land use planning (Johnson et al. 2011; Chetkiewicz and Lintner 2014; BC Ministry of Forests Lands and Natural Resources Operations 2016; Yukon Land Use Planning Council 2018) efforts in Canada. Here we define cumulative effects (CE) as the combined effects of multiple stressors on species or ecosystems over time and (or) space and stressor as all human-induced activities and drivers. We identify two key types of cumulative effects assessments (hereafter CEAs) within Canadian jurisdictions. The first are project-based CEAs (Dubé and Munkittrick 2001; Noble 2010) initiated as part of individual project environmental assessments (hereafter EA). These can include assessments within an Environmental Impact Statement to meet requirements of the *Canadian Environmental Assessment Act 2012* or within an Impact Statement to meet requirements of the *Impact Assessment Act 2019*. The second are effects-based or regional CEAs (Dubé and Munkittrick 2001; Noble 2010) initiated as (*i*) one component of Regional Assessments (*Impact Assessment Act 2019*), which are studies and analyses conducted by an appointed committee or by the Impact Assessment Agency of Canada (IAAC) in areas of existing or proposed projects "to inform planning and management of CEAs and inform project impact assessments" (IAAC 2019; https://www.canada.ca/en/impact-assessment-agency.html), or (*ii*) regional projects or studies conducted by partners or groups within a region of interest (e.g., land use planning region). Both types of CEAs occur within Canadian jurisdictions, but project-based and effects-based CEAs differ in their goals and objectives. We note differences between the science-based goal of most effects-based CEAs (to characterize, analyze, and quantify individual and multiple stressor effects on species and ecosystems within a region) and the more limited goal of project-based CEAs (to undergo project review and approval within a defined regulatory process; Duinker and Greig 2006; Noble 2010; Gunn and Noble 2011; Westwood et al. 2019). Larger, regional scale, effects-based CEAs are needed to quantify the CE of multiple stressors on species and ecosystems where diverse sectors (e.g., in terrestrial systems sectors like forestry, agriculture, energy, mining) and processes (climate change) operate simultaneously (spatially, temporally) within a region. Limitations of project-based CEAs to assess CE at larger regional scales have been widely reviewed (Duinker and Greig 2006; Gunn and Noble 2011; Gillingham et al. 2016; Jones 2016) and include (*i*) insufficient spatial scale and scope to assess a range of stressors; (*ii*) limited resources and data; and (*iii*) uncoordinated, inconsistent, and fragmented individual assessments.

The long history and regulatory requirements of project-based CEA have generated a large body of examples, federal and jurisdictional CEA frameworks, technical guidance documents, and scholarly discussion. Notably lacking, however, is specific scientific guidance that describes and evaluates analytical methods (e.g., qualitative and quantitative analysis methods) that can be used for both project-based and effects-based CEAs where the focus is on terrestrial species and systems. Here qualitative refers to the assignment of a value (e.g., low or high) or ranking for a variable of interest and quantitative refers to the measurement of a quantity for a variable of interest. Regulatory guidance documents and published reviews suggest CEA methods that range from qualitative approaches like questionnaires, interviews, checklists, matrices, and diagrams to quantitative approaches like habitat suitability models, species stressor models, and simulation models (CEAA 2014, 2018; Cocklin et al. 1992; Smit and Spaling 1995; Schultz 2010; Canter and Atkinson 2011; Duinker et al. 2013), but offer little guidance regarding when or why to use specific methods and which CEA questions can be addressed. While this generality and flexibility is understandable given CEA guidance must cover a range of projects, regions, biophysical communities, and valued components (environmental components considered most indicative, critical, or important to ecosystem services or stakeholder values; hereafter VC), it can also give the false impression these methods are equally effective in answering many key science-based CEA questions. For example, one of the most frequently cited rationales for undertaking CEA, in both project-based and effects-based CEAs, is the potential for multiple stressors to create unexpected additive or interactive effects (Didham et al. 2007; Crain et al. 2008; Darling and Côté 2008). However, as many CEA methods are based on existing data or expert opinion, very few are capable of testing for novel or complex effects.

Here our objective is to review and provide guidance on qualitative and quantitative CEA methods for species in terrestrial systems relevant to both project-based CEAs and effects-based CEAs. We focus on terrestrial systems and species because although some methods are applicable to both terrestrial and aquatic systems, some quantitative tools necessary to facilitate CEA methods (e.g., marine spatial planning tools, multi-species models) have been specifically developed for use in marine or aquatic systems. In addition, although recent reviews of CEA methods have been conducted in aquatic systems (see Hodgson and Halpern 2019), no similar reviews have been conducted for terrestrial systems and species. Specifically, we discuss four broad categories of methods with current or potential use for project-based and effects-based CEAs: (*i*) qualitative review, (*ii*) habitat supply models, (*iii*) empirical species–stressor models, and (*iv*) decision support models. We describe each method and provide examples, highlight advantages and limitations, identify how methods address key science-based CEA questions, and provide direction on when and why to use specific CEA methods. The larger scale need for policy shifts and associated mechanisms to promote the implementation of methods reviewed here (i.e., coordinated, collaborative, regional CEAs) are beyond the scope of this document and have been well reviewed elsewhere (e.g., Gillingham et al. 2016; Jones 2016; Hodgson et al. 2019; Westwood et al. 2019).

Our intended audience includes current CEA practitioners; however, we direct this document to biologists, planners, and policy makers new to CEA or considering CEA outside of regulatory project-based CEA. We expect many current practitioners, especially biologists conducting assessments, will be familiar with the methods we outline here. Thus, while we hope our discussion helps consolidate and synthesize information for those individuals, we particularly hope it aids discussions with managers and policy makers to ensure terrestrial CEA frameworks allow for rigorous design and analysis. We structure our discussion/review as follows: (*i*) goals, questions, and objectives of CEA; (*ii*) review of CEA methods; and (*iii*) conclusions and key recommendations.

2. Goals, objectives, and questions related to cumulative effects assessment (CEA)

We acknowledge that goals and objectives for project-based and effects-based CEAs differ because regulatory CEAs specific to Canadian jurisdictions differ from nonregulatory CEAs (Table 1). Project-based CEAs within provincial jurisdictions initiated before

Table 1. Types of cumulative effects assessment (CEA) within Canada.

Type of CEA	Jurisdiction	Agency	Guidance: CEA methods/approach
Project-based CEA			
1. CEA within EA/IA	Provinces	CEAA	Examine cumulative effects (CE) of a project on VCs using five steps: (1) scoping; (2) analysis; (3) mitigation; (4) significance; (5) follow-up (https://www.canada.ca/en/impact-assessment-agency; CEAA 2014, 2018)
		IAAC	Examine CE (changes to environment, health, social, and economic conditions) using five steps preceding (interim guidance; https://www.canada.ca/en/impact-assessment-agency; CEAA 2014, 2018)
	YT	YESAB	Examine CE on VCs by considering project effects (magnitude, duration, timing, likelihood, reversibility, and spatial extent) and other past, current, or future project effects (https://www.yesab.ca; YESAB 2019)
	NT	MVEIRB	Examine CE using four steps: (1) identifying parts of the environment affected by project; (2) determining parts of the environment affected by other past, present, or future projects; (3) predicting effects of proposed project and other projects; (4) identifying ways to manage combined impacts (https://reviewboard.ca; MVEIRB 2004)
	NU	NIRB	Identify and document cumulative impacts from the project and any other past, current, or future project and determine whether project is of regional interest (https://www.nirb.ca; NIRB 2020)
Effects-based CEA			
1. Regional Assessment	Provinces	IAAC	Examine CE using methods identified by the appointed committee or the IAAC and documented in the terms of reference
2. Regional Projects	Any location or jurisdiction	No regulatory agency; collaboration among partners	Examine CE within region of interest using questions and methods defined by partners

Note: Jurisdiction definitions — YT, Yukon Territory; NT, Northwest Territories; NU, Nunavut. Agency definitions — CEAA, Canadian Environmental Assessment Agency; IACC, Impact Assessment Agency of Canada administers projects under *Canadian Environmental Assessment Act 2012* and *Impact Assessment Act 2019*; YESAB, Yukon Environmental and Socio-economic Assessment Board administers projects under *Yukon Environmental and Socio-economic Assessment Act 2003*; MVEIRB, Mackenzie Valley Environmental Impact Review Board administers projects under *Mackenzie Valley Resource Management Act 1998*; NIRB, Nunavut Impact Review Board administers projects under *Nunavut Planning and Project Assessment Act 2013*; partners can be academic institutions, provincial or territorial government agencies, environmental non-governmental agencies, Indigenous communities or groups, research groups, land use planning commissions. Acronyms: EA, environmental assessment; IA, impact assessment; VC, valued components.

28 August 2019 follow technical guidance associated with the *Canadian Environmental Assessment Act 2012* (CEAA 2014, 2018), while those initiated after 28 August 2019 follow technical guidance associated with the *Impact Assessment Act 2019* (IAAC 2019). Critical to the CEAA technical guidance (CEAA 2014, 2018) and operational policy statement (CEAA 2015) is identifying measures that would mitigate adverse CE (step 3) and determining the significance of any adverse CE taking into account the implementation of mitigation measures (step 4). Similarly, new IAAC technical guidance (see https://www.canada.ca/en/impact-assessment-agency/services/policy-guidance/section-22-factors-considered-descriptions.html) requires the consideration of any CE (section 22 (a) (ii)) and any measures that would mitigate any adverse effects of the designated project (section 22 (a) (b)). Project-based CEAs within northern jurisdictions (Yukon Territory, Northwest Territories, Nunavut) follow specific technical guidance (Table 1). The focus of most project-based CEAs in Canada is to identify and mitigate the individual stressors associated with the proposed project. This process involves determining whether project activities, in the context of other past, present, or future activities, will cause significant adverse CE after the implementation of proposed mitigation measures.

Effects-based CEAs conducted as part of Regional Assessments under the IAAC are studies conducted in areas of existing projects or anticipated projects within provincial jurisdictions to inform planning and management of CE and inform project impact assessments. The Minister of Environment and Climate Change appoints a committee or asks the IAAC to conduct a Regional Assessment, establishes the terms of reference for each Regional Assessment, and reviews the final Regional Assessment report. Regional Assessments may differ in their goals, activities, and approaches that can include filling data gaps and analyzing trends, establishing thresholds and standard mitigation, or identifying regional development objectives and scenarios (https://www.canada.ca/en/impact-assessment-agency/services/policy-guidance/regional-assessment-impact-assessment-act.html). We note that Regional Assessments under the IAAC are a new process (two Regional Assessments have been initiated and only one Regional Assessment has been completed as of November 2020). Specific guidance and direction would ensure that future Regional Assessments meet objectives of planning, assessing, and managing CE. Regional Assessments under the IAAC have the potential to be a valuable and useful process for assessing CE in a region using innovative methods, approaches, and tools. We promote the use of rigorous CEA analytical methods within Regional Assessments to (*i*) quantify current and future impacts of human activities and climate change; (*ii*) manage where, when, and how human activity occurs; and (*iii*) produce alternate scenarios to balance human activity and species conservation.

Effects-based CEAs outside of regulatory processes can be conducted in any jurisdiction in Canada by interested partners, groups, or agencies (e.g., academic institutions, provincial or territorial government agencies, environmental nongovernmental agencies, Indigenous communities or groups, research groups, land use planning commissions). The focus of most nonregulatory effects-based CEAs in Canada is often on assessing and analyzing stressor-effect relationships and simulating land and resource development scenarios within a specific region of interest. This typically involves quantifying regional effects of stressors often associated with multiple sectors, identifying types of stressor combinations, measuring uncertainty, and (or) simulating the impacts of stressors on habitats and terrestrial species into the future using various landscape, climate, resource, or management scenarios.

There are multiple CEA definitions, frameworks (see Duinker et al. 2013), processes, and questions (as described previously and in Table 1). In this review, we define key science-based CEA objectives relevant to both project-based and effects-based CEAs as the following: (*i*) describe or characterize overall CE; (*ii*) determine the rank or magnitude and direction (e.g., positive or negative) of individual stressor effects; (*iii*) describe the shape of the relationship between stressors and species (e.g., linear or nonlinear); (*iv*) determine how stressors combine to affect species (e.g., additive, interactive–synergistic, interactive–antagonistic, compensatory, or masking); (*v*) relate CE to baseline or pre-disturbance conditions, defined thresholds, or management triggers; and (*vi*) describe or assign certainty or probability of occurrence of predicted effects. Within our review, we assess the capacity of CEA methods for species in terrestrial systems to address these science-based CEA objectives.

3. Description and evaluation of CEA methods

Here we provide an overview of current methods for project-based and effects-based CEAs that have or could be used for terrestrial species. We conducted a three-stage review to (*i*) identify CEA methods used in project-based CEAs and effects-based CEAs, and (*ii*) document relevant examples. First, we reviewed all active assessments (as of 15 June 2020) listed on the Canadian Impact Assessment Registry (https://iaac.gc.ca) including Environmental Assessments under *Canadian Environmental Assessment Act 2012* (n = 50), Environmental Assessments by Review Panel under *Canadian Environmental Assessment Act 2012* (n = 8), Impact Assessments under *Impact Assessment Act 2019* (n = 2), and Regional Assessments under *Impact Assessment Act 2019* (n = 2) and recorded the types of methods used within the CEA (e.g., Environmental Impact Statement, Impact Statement, Regional Assessment) in Table 2.

Second, we searched for published reviews of CEA methods within the scientific literature and the CEA guidance literature (Table 2). We collapsed similar approaches identified in stages 1 and 2 into four broad categories of CEA methods: qualitative review, habitat supply models, empirical species–stressor models, and decision support models (focusing on Bayesian belief network (BBN) models). We distinguished data products like indicators (i.e., a measurable variable chosen to represent the state of a component) and indices (i.e., composite numerical value for some environmental attribute usually comprised of two or more variables) that can be used within a CEA process from analytical CEA methods. We also distinguished data tools like geographic information systems (GIS), used to produce spatial data products included within most modern project-based or effects-based CEAs, from analytical CEA methods.

Third, we conducted targeted searches using Web of Science across the four CEA methods to find published examples for species in terrestrial systems. We include some CEA published papers as examples in the description of each CEA method in subsequent sections and list all papers reviewed by CEA method including study area location and target terrestrial species or group (Table 3). CEA methods fall on a gradient of increasing complexity, data requirements, and technical expertise. In many cases, a combination of CEA methods will be required to effectively answer science-based CEA questions. Our intent is not an exhaustive review of all CEA methods or an intensive review of one category of methods. Instead, we describe each CEA method and provide relevant examples. We also identify advantages and limitations, explain how methods address science-based CEA questions (Table 4), and provide direction on when and why to use specific CEA methods.

3.1. Qualitative review

3.1.1. Description of method

This broad category of methods includes a variety of approaches to collect (questionnaires, interviews, workshops, literature reviews) and summarize (checklists, matrices, qualitative or conceptual models/diagrams) qualitative data or information (e.g., expert opinion, published or unpublished information) as an exclusive or primary means to assess CE (Cocklin et al. 1992; Smit and Spaling 1995; Duinker et al. 2013; CEAA 2014, 2018; see Tables 2 and 3), rather than direct measurements of species response to habitat change or stressors. This approach would typically include assessments with high constraints on data availability (for species, habitats, and stressors), time, and (or) cost. For example, information on stressors may be based on qualitative descriptions or approximations of development (e.g., hard copies of maps, written or verbal descriptions of proposed developments or stressors), rather than spatial and attribute data on stressors within a GIS. In some cases, qualitative data/information is ranked to provide a categorical assessment of effects (e.g., using literature or expert review to rank/weight potential effects of stressors on species). Formal consensus techniques and qualitative modelling can also be used (e.g., Delphi method, see Linstone and Turoff 2002; fuzzy cognitive mapping, see Gray et al. 2015), although these remain infrequent for most assessments of terrestrial species. Although also a form of qualitative review, we consider independent traditional knowledge (TK) studies or combined TK and scientific knowledge (SK) studies (e.g., co-production of knowledge) to be separate from this category due to distinct methods used to select experts with different types of knowledge (e.g., local harvesters and hunters, Elders, government employees, scientists) and conduct the elicitation procedure (Huntington 1998; Usher 2000; Polfus et al. 2014; Bélisle et al. 2018).

Qualitative review is common among project-based CEAs for terrestrial species (see Table 2, stage 1), where proponents typically conduct literature reviews to collect information and use checklists and matrices to rank potential effects on VC (e.g., low to high). Qualitative review is less common among effects-based CEAs for terrestrial species (outside of TK studies) with the exception of (*i*) potential stressor effects that are difficult to map or quantify (e.g., pesticide use, pollution, species interactions), and (*ii*) complex systems where multiple stressor effects are difficult to visualize or comprehend. In the latter case, conceptual models/diagrams can be combined with other quantitative CEA approaches (e.g., habitat supply, empirical species–stressor, and decision support models) where the conceptual model is shared, refined, and used to guide monitoring and research questions and hypotheses among partners (Marcot et al. 2006; Lindenmayer and Likens 2010; Nelitz et al. 2015).

3.1.2. Examples of applications

We highlight two examples of qualitative review methods and list additional examples from the published literature in Table 3. The first example, from a recent project-based CEA from the IAAC Registry (Environmental Impact Statement for the Marathon PGM-Cu Project 2012), uses a matrix table (Table 5) to identify the VCs, potential effects, mitigation measures, and residual effects of the project. For each VC, gray wolf (*Canis lupus*) and American black bear (*Ursus americanus*), a rank or level is assigned to each parameter used to characterize residual effects (magnitude, spatial extent, frequency, duration, reversibility, ecological or societal value). These levels are then combined to create a measure of overall significance, a required first step in conducting the CEA (CE are defined as the environmental effects resulting from the project in combination with other projects or activities). The second example, from an effects-based CEA, uses a conceptual model/diagram to illustrate the Greater Sage-Grouse (*Centrocercus urophasianus*) annual life cycle including influence of oil and gas development on site fidelity, dispersal, and demography (Fig. 1). The life cycle informs the population simulation model used to track population dynamics and habitat use of simulated Sage-Grouse through space and time as they respond to changing landscape

Table 2. Cumulative effects assessment (CEA) methods identified in the stage one review (Canadian Impact Assessment Registry) and the stage two review (published reviews of CEA methods and publicly available CEA guidance documents).

Stage	CEA methods
Stage 1: Canadian Impact Assessment Registry	**Total: 22/62 (35%) with Environmental Impact Statement, Impact Statement, Regional Assessment Report**
Environmental Assessment under *Canadian Environmental Assessment Act 2012* (n = 50)	**16/50 (32%) with Environmental Impact Statement** 15/16 used QR 4/16 used QR and HSM
Environmental Assessment by Review Panel under *Canadian Environmental Assessment Act 2012* (n = 8)	**5/8 (62.5%) with Environmental Impact Statement** 5/5 used QR 3/5 used QR and HSM 1/5 used QR, HSM, and SM
Impact Assessment under *Impact Assessment Act 2019* (n = 2)	**0/2 (0%) with Impact Statement**
Regional Assessment under *Impact Assessment Act 2019* (n = 2)	**1/2 (50%) with Regional Assessment Report** 1 used QR, HSM, and DSM
Stage 2: Reviews of CEA methods	
Cocklin et al. 1992	Environmental checklists; matrix approaches; input–output methods; network diagram methods
Smit and Spaling 1995	Spatial analysis/mapping; network analysis; biogeographic analysis; interactive matrices; ecological models; expert opinion
Schultz 2010	Habitat suitability models; viability and sensitivity analysis; Bayesian approaches including network analysis
Canter and Atkinson 2011	Indicators and indices; habitat suitability models
Duinker et al. 2013	GIS; scenarios; thresholds; indicators and indices; network analysis; matrices; habitat suitability models/habitat selection models/habitat supply models; stressor-based and effects-based approaches; multivariate statistical analysis; air pollution models; landscape models; animal population simulation models; water-based simulation models; public engagement; community-based monitoring; population viability analysis; cost–benefit analysis; visual amenity analysis; traffic-light decision support system; sustainability appraisal
Hodgson and Halpern 2019	Experiment; meta-analysis; single species models; mapping; qualitative models; multi-species models (aquatic/marine)
Marcot et al. 2001; Marcot et al. 2006; McCann et al. 2006; Uusitalo 2007	Bayesian network models/Bayesian belief network (BBN) models
Stage 2: CEA guidance documents	
Technical Guidance for Assessing CEA under *Canadian Environmental Assessment Act 2012*: Draft (CEAA 2014); Technical Guidance for Assessing CEA under *Canadian Environmental Assessment Act 2012*: Interim Guidance (CEAA 2018)	Questionnaires and interviews; checklists and matrices; network and systems analysis/diagrams; indicators and indices; conceptual and numerical models; trend analysis; spatial analysis
Section 22-Factors to be considered (IAAC 2019; https://www.canada.ca/en/impact-assessment-agency)	New IAAC guidance in preparation; interim guidance is CEAA (2018)
Consideration of cumulative effects in YESAB Assessments (YESAB 2005, 2019)	No CEA methods
Environmental Assessment Guidelines (MVEIRB 2004)	No CEA methods
Proponents Guide: NIRB Technical Guide Series (NIRB 2020)	No CEA methods

Note: Agency definitions: CEAA, Canadian Environmental Assessment Agency; IACC, Impact Assessment Agency of Canada administers projects under *Canadian Environmental Assessment Act 2012* and *Impact Assessment Act 2019*; YESAB, Yukon Environmental and Socio-economic Assessment Board administers projects under *Yukon Environmental and Socio-economic Assessment Act 2003*; MVEIRB, Mackenzie Valley Environmental Impact Review Board administers projects under *Mackenzie Valley Resource Management Act 1998*; Nunavut Impact Review Board (NIRB) administers projects under *Nunavut Planning and Project Assessment Act 2013*. CEA method definitions: QR, qualitative review; HSM, habitat supply models; SM, simulation models; DSM, decision support models.

conditions (climate-induced vegetation change) and exposure to oil and gas stressors (Heinrichs et al. 2019).

3.1.3. Advantages and limitations, CEA questions, and when/why to use

3.1.3.1. Advantages

The advantage of qualitative review is its utility when quantitative data on habitat and stressors and time and (or) financial resources are limited. The approach also has the ability to (*i*) offer a collaborative, flexible, and accessible approach when local issues and values are important (e.g., TK or combined TK and SK studies); (*ii*) identify a wide range of stressors and mechanisms (e.g., indirect effects, species interactions, stressors that cannot be effectively mapped); and (*iii*) serve as a conceptual framework within a CEA using one or more quantitative methods to assess and (or) simulate multiple stressor effects (e.g., empirical species–stressor models; Bayesian belief network (BBN) models; simulation models).

3.1.3.2. Limitations

The limitation of qualitative review is uncertainty surrounding the objectivity and reliability of assessments, particularly when expert opinion is applied to novel locations or stressors. Specifically, these limitations include the inability to quantify any stressor related impacts including effects on habitat area (e.g., amount

Table 3. Cumulative effects assessment (CEA) methods identified in the stage three review (targeted reviews of CEA methods): qualitative review; habitat supply models; empirical species–stressor models; and decision support models (Bayesian belief network (BBN) models).

Stage 3: Targeted reviews of CEA methods	References
Qualitative review	Chilima et al. 2013 (Ontario, CAN; watersheds); Cooper 2011 (United Kingdom; multiple species); Swor and Canter 2011 (Pennsylvania, USA; water, fish, mussels, riparian/floodplain resources)
Habitat supply models	Canter and Atkinson 2011 (USA; multiple species); Houle et al. 2010 (Quebec, CAN; gray wolf); Johnson et al. 2005 (Northwest Territories, CAN; caribou, gray wolf, grizzly bear, wolverine); Mace et al. 1999 (Montana, USA; grizzly bear); Rödder et al. 2016 (Cologne, Germany; Sand Lizard)
Empirical species-stressor models	Sawyer et al. 2009 (Wyoming, USA; mule deer); Houle et al. 2010 (Quebec, CAN; gray wolf); Salice et al. 2011 (South Carolina, USA; Eastern Narrow-mouthed Toad); Salice 2012 (southeast USA; Eastern Narrow-mouthed Toad); Wilson et al. 2013 (northwest Alaska, USA; caribou); Goodale and Milman 2019 (offshore regions; waterfowl, waterbirds, landbirds); Hodgson et al. 2017 (various locations; insects, birds, fish, ungulates); Jones et al. 2017 (Muskoka River Watershed, CAN; benthic invertebrates); Fisher and Burton 2018 (northeast Alberta, CAN; gray wolf, white-tailed deer, moose, American black bear, coyote, Canada lynx, fisher, red fox, snowshoe hare, American red squirrel); Claireau et al. 2019 (western France; 10 bat species); Daniel and Koper 2019 (southern Alberta, CAN; 5 grassland landbird species); Heinrichs et al. 2019 (Wyoming, USA; Greater Sage-Grouse); Mahon et al. 2019 (northeast Alberta, CAN; 27 boreal landbird species); Katzner et al. 2020 (California, USA; roadrunner, hawk); Leston et al. 2020 (northeast Alberta, CAN; 20 boreal landbird species)
Decision support models—Bayesian belief network models	McNay et al. 2006 (north-central British Columbia, CAN; caribou); Steventon et al. 2006 (north coast of British Columbia, CAN; Marbled Murrelet); Howes et al. 2010 (eastern Australia; woodland landbirds); MacCracken et al. 2013 (Bering and Chukchi Seas; Pacific walrus); Ban et al. 2014 (Great Barrier Reef, Australia; reef coral); Fortin et al. 2016 (Alaska, USA; brown bears); Mantyka-Pringle et al. 2016 (South East Queensland, Australia; macroinvertebrates, fish); Mantyka-Pringle et al. 2017 (Slave River Delta, Northwest Territories, CAN; fish, waterfowl, mammals)

of suitable habitat lost) and direct (e.g., habitat loss, mortality) or indirect stressor effects on species (e.g., altered habitat selection, reduced demographic rates like reproduction or survival).

3.1.3.3. Science-based CEA questions
See Table 4 to identify how qualitative review methods address key science-based CEA questions.

3.1.3.4. When/why to use — project-based CEAs
"*When to use*": when quantitative data on species and spatial data on habitats or stressors limit the development of species–habitat models (to assess change in habitat) and (or) species–habitat–stressor models (to assess species response to habitat and stressors). Qualitative data and qualitative review methods can be used to identify, rank, and categorize stressor effects. "*Why to use*": allows for a comprehensive listing and ranking of potential stressor effects on VCs in relation to (*i*) mitigation measures for the proposed project and (*ii*) other past, current, or future projects.

3.1.3.5. When/why to use — effects-based CEAs
"*When to use*": when systems are complex, conceptual diagrams can be combined with other quantitative methods and used to (*i*) identify multiple stressors and (or) pathways of effects on different life stages of target species or VCs (see Fisher and Burton 2018; Pirotta et al. 2018; and Heinrichs et al. 2019 for examples) and (*ii*) outline the analysis process or framework (see Munns 2006 and Katzner et al. 2020 for examples). "*Why to use*": allows for a visual representation of the system (stressors, ecological drivers, individual, and population responses) and the analysis process.

3.2. Habitat supply models

3.2.1. Description of method
In this category, we include various forms of species–habitat modelling, currently the most common approach to terrestrial species assessment in most project-based CEAs as well as in conservation planning and landscape management (e.g., Morrison et al. 2006). Approaches vary in complexity from simple literature-based habitat rankings to complex statistical models of empirical species–habitat relationships (see Table 3). A defining characteristic of this method for CEA is that habitat is used as a surrogate for the identified terrestrial species (or as a VC itself). For habitat supply models, quantitative changes in the amount and (or) quality of a species' habitat (typically categorized and measured within a GIS) are used as the primary measure of CE. In contrast, for empirical species–stressor models, the species direct response to one or more stressors is used as the primary measure of CE. We provide more scientific background to this approach subsequently; however, in general this approach with respect to CEA typically involves:

1. Developing species–habitat associations or models. This requires ranking the suitability/effectiveness/functionality of measurable habitat categories for target species using expert opinion, literature review, and (or) empirical species–habitat correlations.
2. Quantifying changes in the amount of this habitat as a result of the proposed project or projects and existing/future stressors.
3. Inferring effects on species as a result of these habitat changes.

Species–habitat models can be developed using a variety of expert opinion and statistical modelling approaches, have a long history in wildlife management and landscape planning, and are well-documented in the literature (e.g., Morrison et al. 2006; Millspaugh and Thompson 2009; Guisan et al. 2013). Standards for developing literature and expert opinion based habitat suitability indices (HSI) date back to the 1980s (U.S. Fish and Wildlife Service (USFWS) 1980, 1981), and many jurisdictions have similarly developed protocols for wildlife habitat rankings in Canada (BC Resource Inventory Committee 1999; Clarke 2012). These habitat suitability models typically use experts and (or) compile species information from existing literature to rank habitats for a species (e.g., on a 0–1 scale, or similar categorical ranking system). Additional variables that may modify habitat quality (e.g., proximity to roads) are also often considered to differentiate between habitat capability (the potential of the habitat to support the species under optimal conditions) and habitat suitability or effectiveness (the actual ability of the habitat to support the species in a specific study area).

Table 4. Summary comparison of analytical methods for cumulative effects assessment (CEA) for terrestrial species.

	Qualitative review	Habitat supply	Empirical species-stressor models	Decision support models (Bayesian belief networks)
Cumulative effects of stressors based on	• Expert knowledge, literature review of likely effects	• Estimated change in amount of suitable habitat for species as a result of stressors; suitable habitat is characterized using expert opinion/literature and (or) empirical data	• Direct measurements of species' response to gradient of stressors and combinations of stressors	• Modeled response of species abundance or habitat to combined habitat, stressor, and other inputs (inputs may be qualitative or empirical data)
Most commonly associated with	• Project-based CEA	• Project-based CEA, scientific research	• Effects-based CEA, scientific research	• Effects-based CEA, scientific research
Key strengths	• Limited data, time, and cost requirements • Can include a wide range of stressors	• Well established approach • Allows quantification of effects (e.g., loss of suitable habitat) when direct species–stressor response or species data are limited	• Addresses all or most CEA goals/questions using direct measures of species–stressor relationship	• Can incorporate multiple data types and sources (qualitative and quantitative data) • Other strengths/limitations based on the extent to which qualitative versus quantitative data are used
Key limitations	• Fails to answer CEA questions or does so with uncertain reliability	• Incorporates effects only due to habitat loss • Accuracy and reliability highly dependent on quality and precision of species–habitat associations • Fails to address many CEA objectives (interactions, shape of species–stressor response)	• High data, time, and cost requirements • Restricted to species that can be readily sampled	• Infrequent use in terrestrial systems may limit application
Quantitative data requirements				
Species	• **None** — Not quantitative	• **Low** — If based on qualitative species–habitat associations • **High** — If based on empirical data species–habitat associations (see text)	• **Very high** — Requires species metrics across full gradient of stressors (and combinations if testing for interactions)	• Requirements based on the extent to which qualitative versus quantitative data are used in the model
Habitat — spatial data	• **None** — Not quantitative	• **High** — Usually detailed vegetation inventory layers	• **High** — Usually detailed vegetation inventory layers	• Requirements based on the extent to which qualitative versus quantitative data are used
Stressor — spatial data	• **None** — Not quantitative	• **High to very high** — To be effective requires accurate, accessible spatial data on "individual" stressors; often broad sector categories (mining, oil/gas) are used	• **Very high** — To be effective requires accurate, accessible spatial data on "individual" stressors	• Requirements based on the extent to which qualitative versus quantitative data are used

Table 4 (*continued*).

	Qualitative review	Habitat supply	Empirical species-stressor models	Decision support models (Bayesian belief networks)
Ability to answer key CEA questions and application of CEA				
What is the relative magnitude and direction of individual stressor effects?	**X** — Only based on qualitative data	✓ Based on relative contribution of stressor to reducing amount of estimated suitable habitat (requires "backfilling" individual stressors in GIS, see text)	✓✓ Parameter estimates in model represent magnitude, direction and certainty of stressor effects ✓✓ Relative contribution of stressors to loss of suitable habitat can be analysed (requires "back-filling" individual stressors in GIS, see text)	✓✓ Stressor effects typically are estimated by users for input into conditional probability tables (source of input may be qualitative information or empirical models)
What is the shape of the species–stressor response?	**X** — Only via qualitative data	**X** — Not readily accomplished	✓✓ Nonlinear relationships (polynomial) can be tested via model selection and readily graphed	✓✓ Shape of the response can be estimated by users for input into conditional probability tables (source of input may be qualitative information or empirical models)
Do stressors combine additively or interactively?	**X** — Only via qualitative data	**X** — Not readily accomplished	✓✓ Interaction terms can be tested for inclusion via model selection procedures Coefficients/graphs can be used to categorize synergistic versus antagonistic effects	✓✓ Interaction effects can be estimated by users for input into conditional probability tables (source of input may be qualitative information or empirical models)
What is the overall impact of cumulative effects on populations?	**X** — Only based on qualitative data	✓ Based on loss of suitable habitat • Species–habitat model predictions for habitat amounts or species abundance can be compared among "no-disturbance /reference", current, and future landscape conditions, based on GIS/landscape simulation	✓✓ Species–habitat–stressor models can be used to predict and compare species abundance measures among "no-disturbance/reference", current, and future landscape conditions, based on GIS/landscape simulation	✓✓ Population response/output can be compared among "no-disturbance/ reference", current, and future scenarios, with model input (habitat/ stressor amounts) based on GIS/ landscape simulation
What is the overall impact on the community?	**X** — Only based on qualitative data	✓ Requires calculating species–habitat models for all species, calculating predicted abundances in reference, current, or future landscapes, and comparing community metrics among these predicted communities	✓✓ With sufficient data, community metrics can be compared among low–high stressor categories ✓✓ Community metrics can similarly be compared using species abundance in reference, current, or future landscapes, as predicted from empirical species–habitat–stressor models	✓ Requires calculating population responses for species in reference, current, or future landscapes, and comparing community metrics among these predicted communities
How certain are results/ How can certainty be assessed?	**Very low** — Subjective nature makes it difficult to quantify certainty	**Very low** — If species–habitat based on qualitative information (subjective nature makes it difficult to quantify certainty) **Moderate** — If species–habitat relationships based on empirical data	**High** — Statistical significance of stressor effects can be estimated via confidence intervals of parameter estimates	**Moderate to high** — Depends on extent to which models are based on qualitative versus empirical data ✓✓ Approach allows uncertainty of relationships to be quantified and input into the models
Range of stressors evaluated	**High** — Not limited by spatial data on stressors	**Low to moderate** — Limited to stressors with accurate and accessible spatial data	**Low to moderate** — Limited to stressors with accurate and accessible spatial data	**High** — Not necessarily limited by spatial data on stressors if models based on qualitative relationships

Table 4 (*concluded*).

	Qualitative review	Habitat supply	Empirical species-stressor models	Decision support models (Bayesian belief networks)
Ability to incorporate traditional knowledge	**Yes**	**Not usually**	**Not usually**	**Yes**
Statistical/technical expertise required	**Low to moderate** — Moderate with formal modeling	**Moderate**	**High**	**High to very high** (specialized)
Timeframe	**Months**	• **Months to <year** — For species–habitat models based on qualitative data (dependent on status of spatial data, detail of the model, availability of species information) **>Year** — For empirical species–habitat models that require new data collection	• **>Year to multi-year** • Species data collection may need to occur over >1 year to provide sufficient samples • Statistical model development and refinement can be time consuming • Dependent on time required to compile, evaluate, and categorize habitat and stressor data	• **Months to >multi-year** — Highly dependent on source of data input (existing data, new empirical data collection) • Modeling approach may require additional weeks or months

In more recent and complex approaches, empirical species–habitat models are developed by sampling terrestrial species across habitat types in a study area and using advanced statistical analyses to quantify species associations with different habitats. Models are termed resource selection functions (RSF) when species data (presence/absence, used/unused) are analyzed at small, within-population scales and species distribution models (SDM) or species abundance models (SAM) when data (occurrence, abundance, respectively) are analyzed at large, range-level scales (Guisan and Zimmermann 2000, Guisan and Thuiller 2005; Elith and Leathwick 2009; Hegel et al. 2010). For terrestrial birds, current and emerging methods can be followed to convert species counts at a sample station (the most common sample metric for terrestrial birds) to a density estimate (Sólymos et al. 2013; Matsuoka et al. 2014). Reporting species responses as densities (versus relative abundance) provides (*i*) a relevant and readily communicated measure of species response, (*ii*) a standardized measure across regions and studies, and (*iii*) an evaluation against conservation or management targets or population viability estimates.

A range of statistical modeling frameworks are available to analyse species–habitat relationships (De'ath and Fabricius 2000; Austin 2007; Elith et al. 2008; Elith and Leathwick 2009; Zuur et al. 2009; Hegel et al. 2010). Typically, habitat coefficients in these models are used to assign habitat rankings for species. Both expert and empirically based habitat rankings require validation against external data (or internal validations in the case of empirical models) to ensure their reliability for an area (Pearce and Ferrier 2000; Boyce et al. 2002; Muir et al. 2011). Once validated, model results are used to assess the amount and distribution of suitable habitat for species within a GIS (Millspaugh and Thompson 2009). Additionally, when species responses have been converted to density estimates, coefficients can be coupled with GIS maps to estimate species densities or population sizes. Within the context of CEA, the footprint or area of proposed projects, existing projects and disturbances, and future land use scenarios can be overlaid on these maps to estimate the change in suitable habitat and assess effects on terrestrial species based on these habitat changes within the study area.

3.2.2. Examples of applications

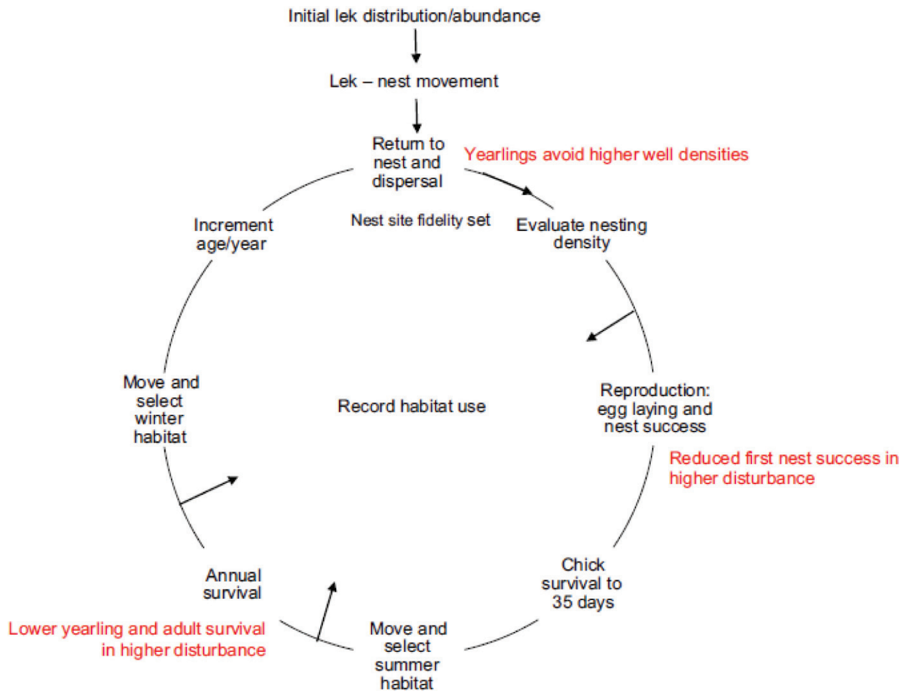
We highlight both project-based CEA and effects-based CEA examples to demonstrate the flexibility of the habitat supply method. Project-based CEAs typically use literature-based or habitat suitability models to identify, map, and quantify the area of suitable habitat for a given species within the study area. We provide a hypothetical example to explain how the habitat supply method is typically used to assess project-specific effects (Table 6). Changes in the amount of suitable habitat during the project (application phase e.g., from 10% to 6% of study area in Table 6) and after the project (closure/remediation phase e.g., 0% change in Table 6) are estimated, relative to current or baseline. The habitat supply method is not typically used to assess CE because the extent to which existing or past stressors have affected pre-existing habitat types is often not known. Instead, CE requirements are often met by comparing the total area of the proposed project (e.g., 5% of study area in Table 6) to the total area of other stressors, regardless of habitat affected (e.g., 35%, Table 6). Thus, in high or moderately disturbed areas, these comparisons may suggest that a project would have minimal added effects relative to existing stressors. Where data and resources allow, some project-based CEAs may use historic maps or create hypothetical no-disturbance landscapes within a GIS (e.g., see methods in ABMI 2017; Mahon et al. 2019) to assess how other stressors may have affected specific habitats. However, resource requirements (time, data, technical expertise) place this type of assessment beyond the mandate or capacity of most project-based CEAs. We include an example of the habitat supply method from a recent project-based CEA from the IAAC Registry

Table 5. Example of a qualitative review (matrix table) for a project-based CEA.

Valued component	Potential effects	Mitigation/ compensation	Residual effect	Significance of residual effects: Magnitude	Spatial extent	Frequency/ probability	Duration	Reversibility	Ecological/ societal value	Overall significance
Gray wolf	Associated with major prey species (moose, deer). At closure increased areas of grassland will create white-tailed deer habitat and potentially higher deer and wolf numbers	Limit footprint of disturbed areas	Increase in deer and wolf populations over time. Residual adverse effect assessed for significance	Study area does not appear to provide high quality moose habitat; therefore, limited wolf population. **MINIMAL**	Loss of habitat limited to site study area. **MINIMAL**	Occurs one or more times, but effect is long-lasting. **MEDIUM**	Loss occurs early in the project and is limited to the phases of the project. **MEDIUM**	Loss of habitat is reversible during site reclamation. Moose followed by wolves should return to site at current levels. **LOW**	Wolves are common in the local and regional study areas. **LOW**	No significant adverse effect. Loss of habitat and population will be limited to the site study area; wolves are mobile and will return after project closure. Effects are reversible through reclamation and habitat restoration
American black bear	Initial migration during site preparation and construction; return to site area after period of human, noise habituation	Limit footprint of disturbed areas. Establish wildlife policy to minimize human interaction with wildlife and decrease potential for habituation	Increased seasonal use of the site over time depends on succession of reclaimed areas. Residual adverse effect assessed for significance	Study area does not appear to provide above average bear habitat. **MINIMAL**	Loss of habitat limited to site study area. **MINIMAL**	Occurs one or more times, but effect is long-lasting. **MEDIUM**	Loss occurs early in the project and is limited to the phases of the project. **MEDIUM**	Loss of habitat reversible during site reclamation. New grassland habitat may increase the seasonal habitat use by bears. **LOW**	Bear density in the study site is likely comparable to that in the surrounding landscape. **LOW**	No significant adverse effect. Loss of habitat and population will be limited to the site study area; bears are mobile and will return after project closure. Effects are reversible through reclamation and habitat restoration

Note: The table summarizes the proposed project effects; mitigation and compensation measures; potential residual effects (the environmental effect of a project that remains after mitigation measures have been implemented); significance of residual effects in consideration of a variety of assessment factors (ratings in boldface type); and whether a significant adverse environmental effect on a valued component is expected as a result of project implementation. Source: Environmental Impact Statement for the Marathon PGM-Cu Project 2012. The table represents descriptions and ratings summarizing the environmental effects of a metal mine development on two valued components (gray wolf and American black bear).

Fig. 1. Example of a qualitative review (conceptual model/diagram) for an effects-based CEA. Figure represents an annual life cycle of events of Greater Sage-Grouse responding to oil and gas development in southwest Wyoming, USA (from Heinrichs et al. 2019, reproduced with permission of Ecol. Appl., Vol. 29, ©2019 Ecological Society of America).



(Environmental Impact Statement for the Springbank Off-stream Reservoir Project 2018) to estimate area changes in the amount of suitable habitat (based on habitat suitability models) for select VCs Sprague's Pipit (*Anthus spragueii*) and Northern Leopard Frog (*Lithobates pipiens*) during different flood scenarios created by water diversion to a channel and reservoir within the proposed project area (Table 7).

Effects-based CEAs typically involve the creation and application of empirical species–habitat models within the study area. Mace et al. (1999) evaluated the CE of human and resource activity on grizzly bear (*Ursus arctos*) habitat in western Montana during spring, summer, and fall seasons. The authors modelled bear resource selection from telemetry data, satellite imagery, elevation, human activity points, roads, and trails and extrapolated models to a larger region with similar climate and vegetation. Quantifying and mapping areas with a high probability of use by bears in each season was used to inform conservation and management actions (e.g., road or trail closures within important seasonal habitats, habitat restoration, future road creation). Rödder et al. (2016) outline an innovative approach to integrate SDM and potential connectivity models to identify connected and isolated populations and habitat corridors for a species of conservation concern, the Sand Lizard (*Lacerta agilis*), in western Germany (Fig. 2). Quantifying potential connectivity areas (for movement like dispersal or migration) was used to identify areas of strong connectivity, persistent isolation, and stable connective networks needed to connect populations. This proposed framework may be particularly useful for proposed developments that affect (*i*) portions of a previously larger, interconnected population, or (*ii*) connective features between permanently colonized habitat patches. Additional examples from the published literature are listed in Table 3.

3.2.3. Advantages and limitations, CEA questions, and when/why to use

3.2.3.1. Advantages

Advantages of the habitat supply method include its ability to quantify effects using loss of suitable habitat. This approach, which has wide-spread application in wildlife research, may be suitable when (*i*) data to map and measure habitat change are available, but empirical data on species response to specific stressors are limited; (*ii*) collaborative approaches focused on local issues and values are important (e.g., species–habitat associations may be based on both qualitative (TK, local and expert knowledge) and quantitative (species–habitat models) data); and (*iii*) data to develop empirical species–habitat models and adjusted density estimates are available so that total impacts can be compared to current and future population targets (e.g., Mahon et al. 2014).

3.2.3.2. Limitations

Limitations to the habitat supply modelling approach include the availability of accurate, publicly available spatial data for habitats and stressors, and reliable species−habitat associations from the region of interest. If additional data are required to refine existing habitat suitability models or create new empirical species–habitat models within a study area, stakeholders or partners should a priori develop rigorous sampling designs and survey protocols to meet project objectives. We note two additional cautions when using this approach. Species−habitat associations based on limited empirical data from within the region or data from outside the region require model validation and testing (Muir et al. 2011). Finally, stressor effects are limited to how stressors contribute to the loss of suitable habitat (habitat changes are used to infer population consequences). Habitat-based analysis alone may not be useful because (*i*) habitat is assumed to be a useful measure of population status, (*ii*) habitat needs of the target species or VC are assumed to be known (Schultz 2010), and (*iii*) specific stressor effects on the target species or VC are not quantified (e.g., the shape, magnitude, and direction of species' response to stressors; indirect or novel effects of stressors on species; or interactions among stressors). Failure to identify which stressors are influencing target species or VCs may limit future research into biological mechanisms (e.g., habitat selection, movement, dispersal, reproduction, survival, predator–prey dynamics) needed to

Table 6. Example of habitat supply models for a project-based CEA. In this hypothetical 10 000 ha study area, a proposed project is estimated to have a 500 ha development footprint, most of which (400 ha) falls on the existing 1000 ha of suitable habitat for a species.

	Baseline (current conditions)		Project application		Project closure (with remediation)	
	Area (ha)	Percent study area (%)	Area (ha)	Percent study area (%)	Area (ha)	Percent study area (%)
Species suitable habitat	1000	10	600	6	1000	10
Undisturbed habitat	6500	65	6000	60	6450	64.5
Disturbance						
Stressor A	1000	10	1000	10		
Stressor B	2000	20	2000	20		
Stressor C	500	5	500	5		
Total existing stressors	3500	35	3500	35	3500	
Proposed project			500	5	50	0.5
Total disturbance	3500	35	4000	40	3550	35.5

Note: Project start/operation (i.e., application) and end (i.e., closure) conditions are compared to current (i.e., baseline) conditions to estimate project-specific effects on species (suitable habitat is reduced from 10% to 6% of study area during the project, but 0% after remediation and closure). Cumulative effects are typically evaluated by summing the total area of existing stressors (35% of study area) and comparing to the area of the proposed project footprint (5%) to indicate the project's relative or incremental effect.

understand species responses and develop effective management or mitigation actions.

3.2.3.3. Science-based CEA questions

See Table 4 to identify how habitat supply model methods address key science-based CEA questions.

3.2.3.4 When/why to use – project-based and effects-based CEAs

"*When to use*": when species data are limited, expert opinion habitat suitability models can be used to quantify the loss of suitable habitat. When species data are available, but data to model the response of species to key stressors are limited, empirical species–habitat models can be used to quantify loss of suitable habitat. Models can be extrapolated to larger regions to compare the current total cumulative loss of suitable habitat within proposed project areas and larger regional areas (e.g., planning units or ecological/biological regional units). Newly available environmental data (e.g., sentinel missions) could support species–habitat model development at finer spatial and temporal scales. "*Why to use*": despite limitations and assumptions, these flexible and diverse methods can allow for a comprehensive assessment of changes in the amount of suitable habitat for multiple terrestrial species during each stage of industrial or resource development (e.g., before and after development, after reclamation or restoration) within a proposed project and regional study area (e.g., project-based CEA) or larger regional study area (e.g., effects-based CEA).

3.3. Empirical species-stressor models

3.3.1. Description of method

This method combines carefully designed, large-scale field studies and advanced statistical modelling to directly measure and quantify species responses to different stressor amounts and combinations (rather than inferring effects based on changes in the amount of suitable habitat alone). Although some elements are common to empirical species–habitat models, stressors and potential stressor interactions are the primary factors of interest in design and analyses of empirical species–stressor models. Quantifying the individual and combined effects of stressors on populations and identifying their temporal and spatial scales of influence are key challenges in assessing CE and planning future landscapes that support populations of terrestrial species (Foley et al. 2017; Heinrichs et al. 2019). Once stressor effects are identified, additional work to identify changes in ecological and behavioural processes (e.g., reproduction, survival, habitat selection, predator, and parasite dynamics) can be used to develop and evaluate effective mitigation or management actions. The approach requires (*i*) targeted sampling of species metrics across a full gradient of stressor amounts, stressor combinations, and habitats and (*ii*) advanced design and interpretation of statistical models to determine the shape, magnitude, direction, and additive or interactive effects of multiple stressors. The approach is sometimes described as a dose–response approach, although logistical and resource data constraints have limited its application for many terrestrial species. We provide additional scientific background subsequently; however, for CEA this approach generally involves three key steps:

1. Assessing stressors and availability of adequately attributed stressor data.
2. Developing a model set and sampling design to ensure adequate sampling of key stressor combinations and gradients.
3. Conducting statistical analyses and interpreting the often-complex results to provide land use and mitigation recommendations, summarize data gaps, and prioritize future research and monitoring needs.

Further we outline key design and analytical steps of the approach, focusing on technical considerations that are unique and (or) go beyond those typically used in empirical species–habitat models. First, in addition to spatial data commonly used for species–habitat models (e.g., landcover, vegetation, other environmental variables), the approach requires compiling and assessing the accuracy of attributed stressor layers (e.g., including information on originating sector, type of stressor, year of origin and removal, reclamation status, buffer width). Second, GIS analyses are used to summarize and categorize stressor types, amounts, and degree of co-occurrence in the study area. This information can be used to develop specific a priori hypotheses on how stressors combine (e.g., additively or interactively) to affect species in the study area (often represented as model sets that are later evaluated in statistical analyses; see Table 8 as an example). Stressor and habitat layers are queried to create a targeted sampling design that ensures sufficient species data can be collected across the full range of stressor amounts and combinations (approximating a "full factorial" design to the extent possible).

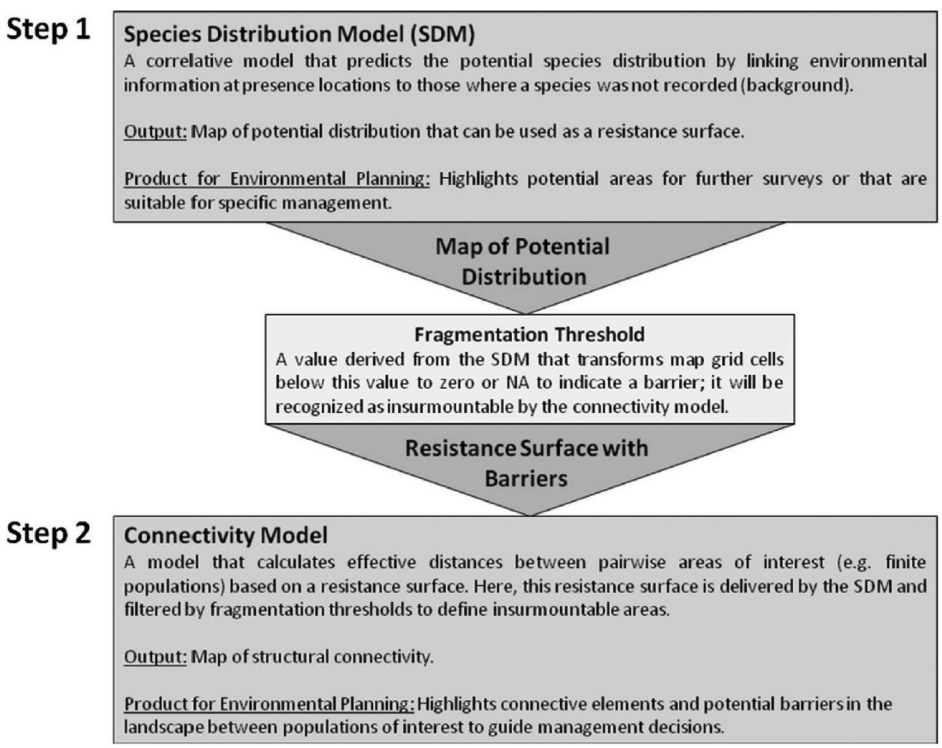
Given the large data requirements of species–habitat–stressor models, existing species data from previous assessments, research, or monitoring may be assessed for potential compatibility and integration with data from targeted sampling designs (see Foster et al. 2017 for an example approach). The large resource investment of the

Table 7. Example of habitat supply models for a project-based CEA.

						Change from baseline					
						Design flood		1:100 year flood		1:10 year flood	
Valued components	Habitat suitability rating	Baseline (ha)	Design flood (ha)	1:100 year flood (ha)	1:10 year flood (ha)	ha	%	ha	%	ha	%
Sprague's Pipit	High	0	0	0	0	0	0	0	0	0	0
	Moderate	0	0	0	0	0	0	0	0	0	0
	Low	817.7	621.8	731.7	814.2	–195.9	–24.0	–86.0	–10.5	–3.5	–0.4
	Very low to nil	4042.2	4238.1	4128.2	4045.7	195.9	4.8	86.0	2.1	3.5	0.1
Northern Leopard Frog	High	49.6	36.6	38.9	49.6	–13.0	–26.2	–10.7	–221.6	0	0
	Moderate	93.7	90.8	98.0	93.7	–2.9	–3.1	4.3	4.6	0	0
	Low	192.0	618.9	462.0	195.8	426.8	222.3	270.0	140.6	3.8	2.0
	Very low to nil	4524.6	4113.6	4261.0	4520.8	–411.0	–9.1	–263.6	–5.8	–3.8	–0.1

Note: The table represents changes in habitat area for multiple flood scenarios based on habitat suitability models for two valued components (Sprague's Pipit and Northern Leopard Frog). Source: Environmental Impact Statement for the Springbank Off-stream Reservoir Project 2018.

Fig. 2. Example of habitat supply models for an effects-based CEA. Figure represents a framework to quantify potential dispersal corridors for sensitive species by integrating a species distribution model with a connectivity model (from Rödder et al. 2016, reproduced with permission of Environ. Manage., Vol. 58, ©2016 Springer). Notes — Resistance surface: the species distribution map can be used as a resistance surface where high suitability values indicate low resistance values after accounting for barriers (i.e., fragmentation thresholds). Fragmentation thresholds: barriers that limit unrealistic movement paths in the landscape (e.g., disturbed area or large water body with low suitability values that cannot be crossed by the species). Structural connectivity: spatial arrangement of landscape elements.



approach makes it particularly critical that new data collection adhere to existing species sampling standards.

Finally, as with empirical species–habitat modelling, a range of statistical modeling frameworks are available to analyse species–habitat–stressor relationships (De'ath and Fabricius 2000, Austin 2007; Elith et al. 2008; Elith and Leathwick 2009; Zuur et al. 2009; Hegel et al. 2010). For example, generalized linear models and associated methods are commonly used in ecological research, and model coefficients can be readily interpreted and communicated. For project-based or effects-based CEAs, we recommend a two-step modelling approach to (*i*) determine the best species–habitat model (a "baseline" model that includes habitat, landscape, and sampling metrics already known/expected to affect species abundance) and (*ii*) test whether the addition of stressor variables improves model fit and explains abundance patterns above habitat loss alone. Additional comparisons of paired additive (A + B) and interactive (A + B + A × B) models within the model set can assess if simple additive or more complex interactive effects may be occurring (e.g., Mahon et al. 2019). Interactive effects could be synergistic (the interaction of stressors increases the magnitude of the response above that seen for additive effects; A + B + A × B > A + B) or antagonistic (the interaction of stressors decreases the magnitude of the response below that seen for additive effects; A + B + A × B < A + B). As in species–habitat models, the predictive ability of models can be evaluated using external data, although internal validation approaches (e.g., cross-

Table 8. Example of empirical species–stressor models for an effects-based CEA.

Model group and set	Model type	Model predictors
Habitat only (HO)		
1a	N/A	Habitat and spatial variables only
Linear (LI)		
2a	Additive	NL + WL
2b	Interactive	NL + WL + NL × WL
Linear and energy (LE)		
3a	Additive	NL + WL + WE
3b	Interactive	NL + WL + WE + WE × NL + WE × WL
3c	Interactive	NL + WL + WE + WE × NL + WE × WL + WE × NL × WL
Linear and forestry (LF)		
4a	Additive	AL + HU
4b	Interactive	AL + HU + HU × AL
Cumulative (CU)		
5a	Additive	AL + WE + HU
5b	Interactive	AL + WE + HU + HU × AL + HU × WE
5c	Interactive	AL + WE + HU + HU × AL + HU × WE + WE × AL + HU × WE × AL
Disturbance area (DA)		
6a	Additive	All disturbances summed (combined AL, WE, HU)

Note: All models include habitat and spatial predictors selected for each landbird species. Stressor definitions: NL, narrow linear (percent cover of seismic lines); WL, wide linear (percent cover of pipelines, powerlines, all roads); WE, wells (percent cover of bitumen, oil, and natural gas wells); AL, all linear (percent cover of narrow and wide linear combined); HU, harvest unit (percent cover of harvest units with harvest activity <20 years). Candidate model set represents an approach to test additive and interactive effects of multiple sectors and stressors on boreal landbirds within the Athabasca Oil Sands Area of Alberta, Canada. Source: Mahon et al. 2019.

validation methods) will likely be more common in this particularly data intense approach (Pearce and Ferrier 2000; Boyce et al. 2002). As in empirical habitat supply approaches, final model habitat and stressor coefficients for each species can be combined with study area maps within GIS to evaluate species distribution and abundance given stressor conditions. However, it is stressor amounts and stressor combinations (additive or interactive), not just habitat amounts, that are used in these analyses. Species–habitat–stressor models can also be combined or integrated with various landscape simulation models to predict future species abundance or density under alternate resource and land management, natural disturbance (wildfire, floods, drought, insect outbreaks), and climate change scenarios.

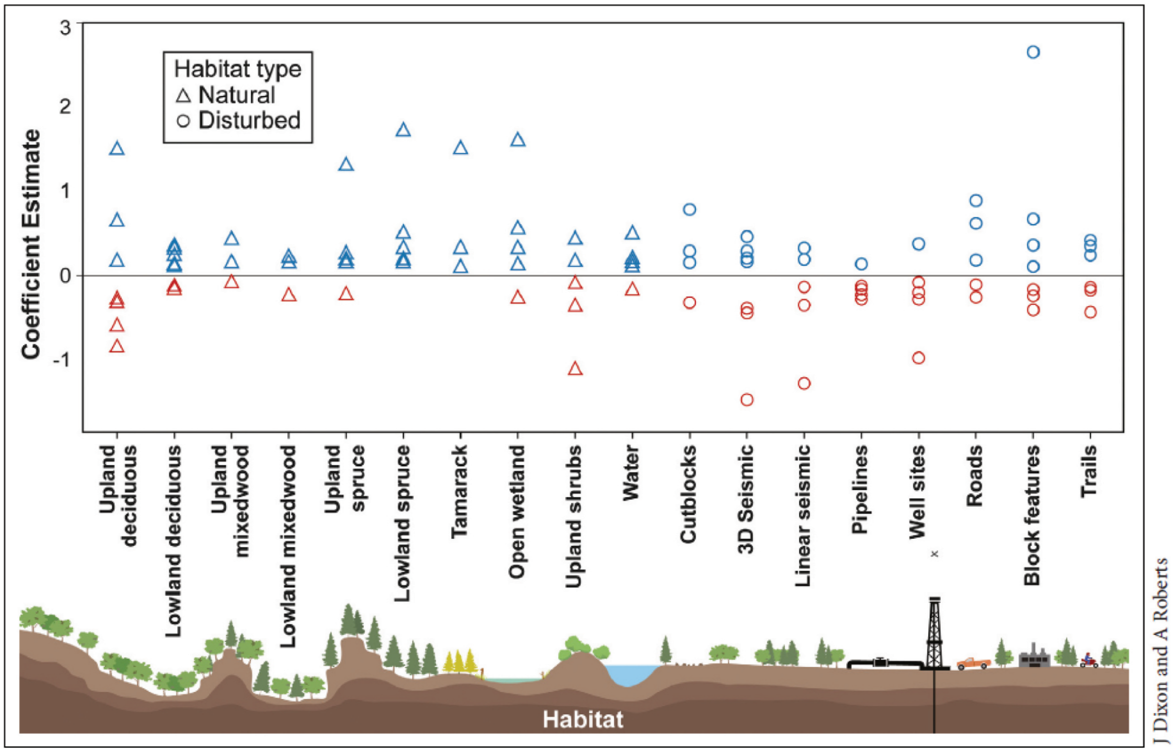
3.3.2. Examples of applications

We provide a number of examples to highlight the diversity of analysis approaches within this broad category. We focus on examples that include multiple terrestrial species, clearly defined objectives, and innovative and diverse sampling and analysis methods. Recent studies from northern Alberta, Canada, a region where multiple resource sectors (forestry, agriculture, conventional oil and gas exploration and development, oilsands exploration and development) and intensive land use practices create co-occurring stressors across time and space, have used versions of empirical species–stressor modelling to assess CE on wildlife including boreal birds (Schieck et al. 2014; Sólymos et al. 2015) and mammals (Fisher and Burton 2018). We highlight the latter study as an innovative assessment using digital cameras to assess effects of polygonal (harvest units, well sites, industrial sites, mines) and linear stressors (seismic lines, roads, trails) on 10 terrestrial mammal species: gray wolf, white-tailed deer (*Odocoileus virginianus*), moose (*Alces alces*), American black bear, coyote (*Canis latrans*), Canada lynx (*Lynx canadensis*), fisher (*Pekania pennanti*), red fox (*Vulpes vulpes*), snowshoe hare (*Lepus americanus*), and American red squirrel (*Tamiasciurus hudsonicus*). The authors use model coefficients to quantify the magnitude, direction, and statistical certainty of individual stressor effects on mammal species relative abundance to demonstrate the variability among species and stressors in an effective, easy to interpret format (Fig. 3).

Despite growing evidence of interactive effects as key factors structuring terrestrial and marine communities (Brown et al. 2013; Cartwright et al. 2014), few studies have attempted to examine these complex relationships in terrestrial systems. Mahon et al. (2019) examined additive and interactive effects of multiple polygonal and linear stressors on boreal landbirds in northern Alberta. The authors evaluated whether the effects of multiple stressors combined additively or interactively by testing a candidate model set of 12 paired CE models of abundance for 27 species of boreal landbirds (Table 8). Creating species–habitat and species–habitat–stressor models revealed that the addition of disturbance effects significantly improved models for 20 of the 27 (74%) landbird species. Complex synergistic and antagonistic interactions among stressors were observed for 40% of landbird species, with the majority of interactions being synergistic, suggesting that both direct effects (e.g., loss of native vegetation) and indirect effects (e.g., altered biological interactions, changes to behaviour or vital rates) are affecting landbird populations within the regional study area.

A number of studies have used a zone of influence (ZOI) approach to assess how distance from stressors can influence the magnitude of the stressor response for multiple terrestrial species (e.g., Claireau et al. 2019; Daniel and Koper 2019). These studies can be useful to directly inform mitigation or management actions. Careful study designs should attempt to (*i*) examine species responses to all primary stressors, (*ii*) include distance to stressor as a continuous variable, (*iii*) avoid pre-defined zones, (*iv*) use multiple response metrics (e.g., abundance, movement, or demographic variables), and (*v*) assess underlying biological processes (e.g., habitat selection, predator–prey interactions) to avoid potentially misleading results and oversimplified or misguided management actions. Daniel and Koper (2019) estimated effects of distance from energy-related infrastructure (oil wells, shallow gas wells, roads) and shallow gas well density on habitat use (abundance used as an index of habitat use) and productivity (nest success, clutch size) of five grassland songbird species, the Chestnut-collared Longspur (*Calcarius ornatus*), Sprague's Pipit, Savannah Sparrow (*Passerculus sanwichensis*), Vesper Sparrow (*Pooecetes gramineus*), and Western Meadowlark (*Sturnella neglecta*). The authors assessed CE of all energy infrastructure on habitat use

Fig. 3. Example of empirical species–stressor models for an effects-based CEA. Figure represents estimated parameter coefficients (effect sizes) for the relationship between species abundance and landscape features (natural, disturbed), from species distribution models (SDM). Some species relationships were positive (blue) and others were negative (red) for both natural features (triangles) and disturbance features (circles) (from Fisher and Burton 2018, reproduced with permission of Front. Ecol. Environ., Vol. 16, ©2018 Ecological Society of America). Notes — Mean effect sizes of disturbance stressors on species distribution were equal or greater than those of natural land cover variables. The direction and magnitude of these effects varied across species and stressor types (four species showed strong positive associations with industrial block features and three species showed strong negative associations with 3D seismic lines). Stressor definitions: cutblock, forest harvest block; 3D seismic, intensively clustered, crossed three-dimensional seismic lines; linear seismic, long linear petroleum exploration lines; pipeline, oil or gas pipeline; well site, small square associated with oil or gas exploration; road, hard surface road; block feature, polygonal industrial sites; trails, soft surface trail.



and productivity by comparing two types of impact models: single strongest effect (e.g., strongest impact) and average effect (e.g., additive impact) models. As a final predictive map product for each bird species, they used their model results (including threshold distances) to estimate the species-specific cumulative area within the study area that was likely to have poor, low, medium, or high habitat potential for abundances, clutch sizes, or nesting success.

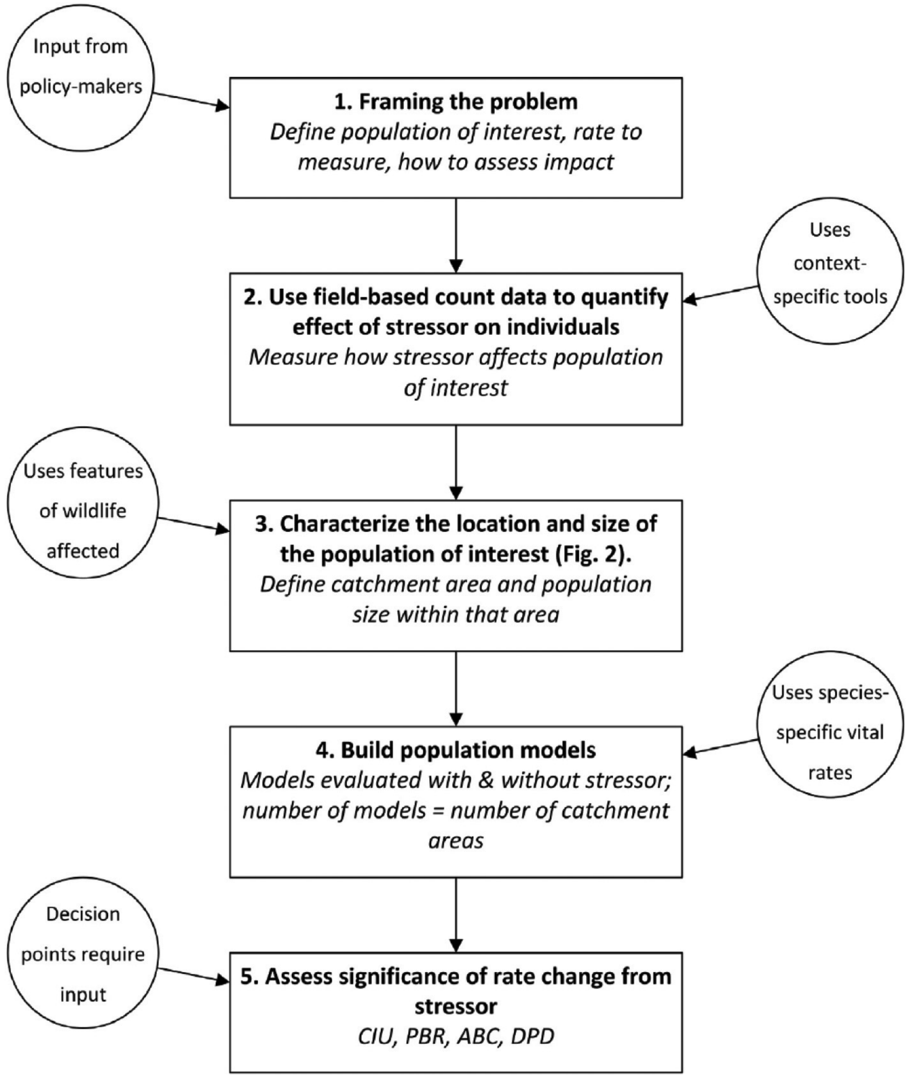
Finally, a number of studies have used population models to integrate life history (e.g., demographic parameters), stressor effects, and environmental effects (Salice et al. 2011; Salice 2012; Heinrichs et al. 2019) to better understand how stressors acting on different life stages will affect population-level rates of change. Hodgson et al. (2017) and Katzner et al. (2020) provide useful frameworks to assess population level consequences of stressors on terrestrial and aquatic species. The five-step framework (Fig. 4) proposed by Katzner et al. (2020) may be particularly useful to upscale from individual-level effects to population-level consequences (i.e., from count-based data to rate-based estimates describing the larger population). This type of framework is critical to (*i*) interpreting the relevance of models describing the consequence of stressors on wildlife populations (e.g., assessing the significance to the population of the rate changes brought on by the stressor or stressors) and (*ii*) developing effective conservation or management actions (e.g., threshold for amount of anthropogenic take). The authors provide relevant examples for two terrestrial birds: Greater Roadrunner (*Geococcyx californianus*) affected by solar energy in the Mojave Desert and Red-tailed Hawks (*Buteo jamaicensis*) affected by wind turbines in central California. Additional examples from the published literature are listed in Table 3.

3.3.3. Advantages and limitations, CEA questions, and when/why to use

3.3.3.1. Advantages

Advantages to empirical species–stressor modelling are that species responses to stressors are measured directly (not inferred based on changes to suitable habitat). Measuring and analyzing species responses across a full combination and gradient of stressors is the only way to achieve many of the key science-based objectives of CEA and quantitatively evaluate (*i*) individual stressor effects and an associated measure of certainty of the effect; (*ii*) additive and interactive effects; and (*iii*) shape, magnitude, and direction of species–stressor relationships. This approach can be used to identify the presence of complex indirect effects beyond habitat loss alone and focus new research on ecological and behavioural processes, such as shifts to alternate food or prey and changes in reproduction or survival, habitat selection, and predator or parasite dynamics. Recent work on birds and mammals from multi-stressor landscapes in the western boreal forest demonstrates the diversity and complexity of species responses to individual and multiple stressors (see Sólymos et al. 2015; Fisher and Burton 2018; Mahon et al. 2019) and identifies the need for process and whole system studies to develop biologically relevant and effective mitigation or management actions.

Fig. 4. Example of an empirical species–stressor model framework (integrated population model) for an effects-based CEA. Figure represents a framework to assess population-level consequences of human stressors on wildlife. Square boxes are steps in the framework; circles are outside information required to process each step. From Katzner et al. (2020), reproduced (free of copyright) from Ecosphere, Vol. 11, ©2020 Ecological Society of America. Notes — Approaches to understanding the demographic significance of human impacts include: CIU, counterfactual of impacted and unimpacted population (ratio-based comparison of scenarios derived from population models with and without the human stressor); PBR, potential biological removal (fixed harvest rate applied to a population while allowing the population to reach or maintain an optimum); ABC, acceptable biological change (estimate of the amount of human-associated take that results in a 33% probability of a defined population target being achieved); DPD, decline probability difference (probability that a population would decline in scenarios with and without a single human stressor).



Finally, sampling for these large-scale studies is typically designed to capture a large number of species, which allows for a proactive and robust assessment of stressor effects on both individual species and the community (see Mahon et al. 2019 for an example). For species at risk or species of high conservation concern, data intensive integrated population models can be created to better understand when (e.g., life stage) and how (e.g., reproduction, dispersal, survival) individual and multiple stressors influence populations (see Heinrichs et al. 2019 for an example and Katzner et al. 2020 for a framework).

3.3.3.2. Limitations

Limitations of this approach include intensive field projects (time, resources) to collect data across the full gradient of stressors, stressor combinations, and habitats; technical expertise to conduct modelling; and availability of accurate and sufficiently attributed spatial data. In addition, two modelling restrictions also limit this approach: (*i*) stressors and stressor combinations (additive, interactive) can only be modelled if they co-occur within the study area along a gradient from low to high intensity and can be sufficiently sampled and (*ii*) rare, specialized, and patchily distributed species can only be modelled if sample size requirements for complex statistical models are met. This may include some, but not all, species of conservation or management concern (e.g., species listed "at risk" under the *Species at Risk Act*, rare, or specialist species). Including a large suite of species can be used to proactively assess species and community shifts, including the biotic homogenization of communities as a result of human disturbance (Clavel et al. 2011; Mahon et al. 2016, 2019).

3.3.3.3. Science-based CEA questions

See Table 4 to identify how empirical species–stressor model methods address key science-based CEA questions.

3.3.3.4. When/why to use – project-based and effects-based CEAs

"*When to use*": when species and stressor data are available or can be collected, empirical species–habitat–stressor models can be used to address key science-based CEA questions. Models can be used to assess community shifts or population level changes between historical or no-disturbance and current landscapes (Mahon et al. 2019). Empirical species–habitat–stressor models can also be applied to landscape simulation models to generate realistic scenarios about future impacts of stressors, climate, and natural disturbance regimes on species of interest. When resources are available (e.g., for high profile species at risk), we recommend integrated population modelling frameworks and approaches (see Katzner et al. 2020 for a framework and examples). "*Why to use*": despite large data requirements, these diverse methods can quantify direct (additive effects of loss of suitable habitat) and in some cases indirect effects (synergistic or antagonistic effects caused by complex interactions) of multiple stressors on terrestrial species. Additional demographic studies to identify how and when stressors affect specific life stages and vital rates (e.g., reproductive output, dispersal rate, survival or mortality, population growth rate, probability of extinction) are key to developing integrated population models needed to (*i*) assess the significance to the population of the rate changes brought on by the stressor or stressors, (*ii*) develop biologically relevant thresholds for anthropogenic or human-caused take, and (*iii*) devise targeted and effective mitigation measures.

3.4. Decision support models

3.4.1. Description of method

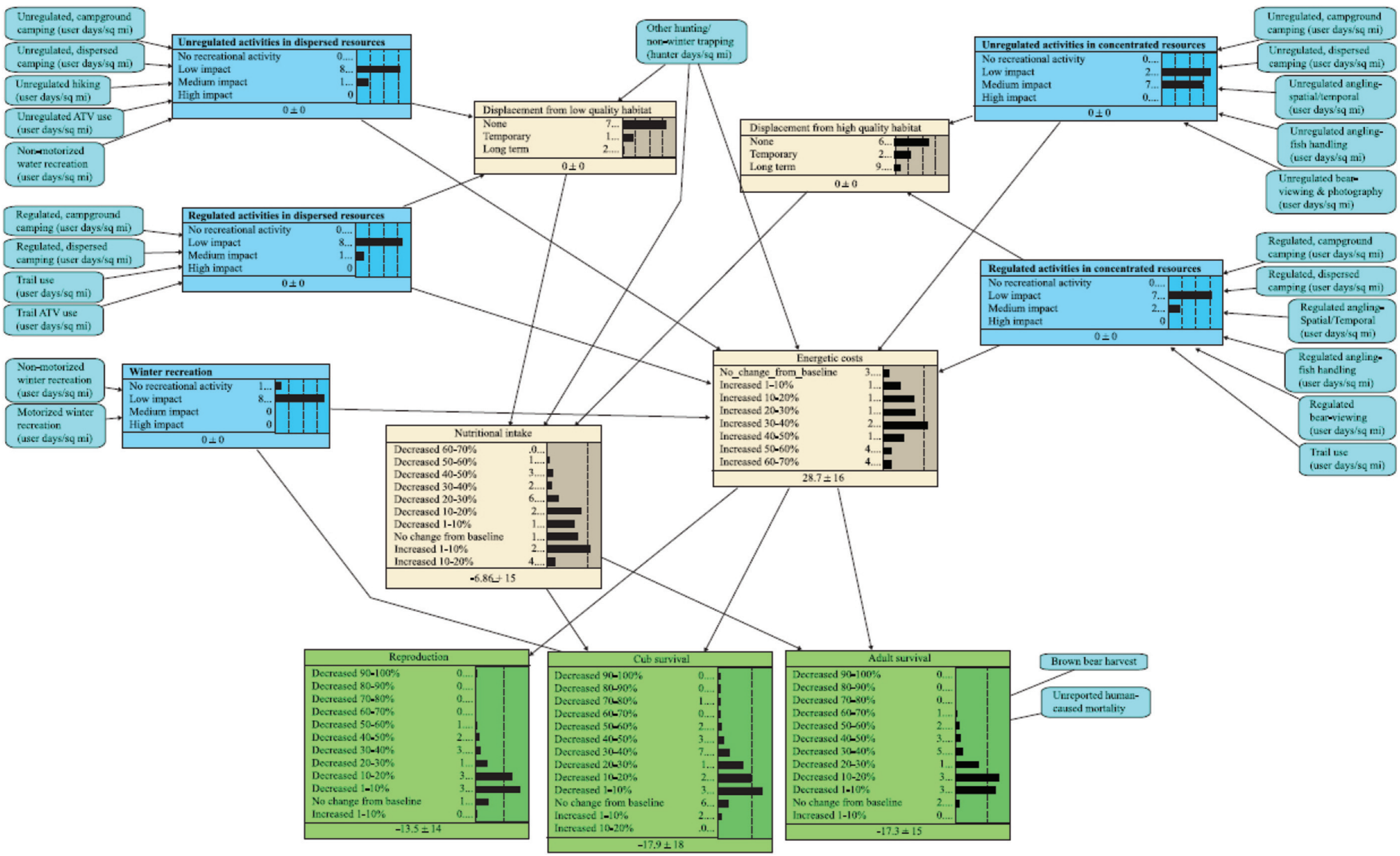
For this category, we restrict our review of decision support models to BBN models (also called belief networks, probability networks, Bayesian networks, and Bayes nets). Use of BBNs in the ecological sciences has expanded over the past two decades (Marcot et al. 2001; Steventon et al. 2006; Howes et al. 2010; MacCracken et al. 2013; Ban et al. 2014; Fortin et al. 2016; Mantyka-Pringle et al. 2017) and descriptions and guidelines for their development and use have been published (Marcot et al. 2006; McCann et al. 2006; Uusitalo 2007; Jensen and Nielsen 2007; Kragt 2009). In general, BBN models allow users to graphically and quantitatively represent cause–effect relationships among ecosystem components (e.g., stressors, habitats, species) using a range of data types and sources (i.e., qualitative and quantitative information). Technically, BBN models consist of three elements: (*i*) nodes representing key predictor or explanatory variables and one or more response variables; (*ii*) links between nodes that represent cause–effect relationships; and (*iii*) joint probabilities representing the belief that node states will have certain probabilities given the probabilities of the states of connected nodes (MacCracken et al. 2013). Nodes represent only categorical variables; continuous variables must be divided into discrete categories or states. Nodes that do not have links to other nodes that influence their prior probabilities are parent (independent) nodes. Nodes that have links to other nodes that influence their prior probabilities are child (dependent) nodes. Child nodes with no outgoing links are output nodes. Each node contains a conditional probability table that specifies the relationships among all combinations of the states of the parent node and the states of the child node where all probabilities sum to one. Child nodes with more than one link represent interactions among parent nodes and capture the CE of multiple variables. The value for each cell in a conditional probability table can be derived from empirical data or an expert elicitation process that requires experts to document their certainty about the relationships. The final posterior probabilities of values or states of the output nodes are calculated in the network using standard Bayesian learning statistics (Spiegelhalter et al. 1993). The algorithms in commercial BBN modelling platforms (e.g., Hugin: http://www.hugin.com; Netica: http://www.norsys.com; SamIam: http://reasoning.cs.ucla.edu/samiam/; B-Course: http://b-course.hiit.fi/; see Uusitalo 2007) permit rapid updating of probabilities throughout the network as data and evidence become available.

A BBN modelling approach can be used within CEA studies to effectively support management decisions by (*i*) graphically presenting the complexity of the ecosystem in a hierarchy that partitions the problem into solvable steps; (*ii*) accommodating qualitative opinions (e.g., costs, benefits, and uncertainties) from a variety of partners, combining them with quantitative data, and then incorporating all information into a formal evaluation of outcomes given specific actions; and (*iii*) representing system variability and uncertainty and their implications to management actions using probabilities (McCann et al. 2006). BBN models provide flexible frameworks to assess multiple stressor effects using quantitative and qualitative information (including TK and SK information; see Bélisle et al. 2018 for an example). Note that to meet many CEA objectives (e.g., shape, magnitude, and direction of stressor effects; additive or interactive stressor combinations), BBN models will need to include quantitative data (e.g., cell values in conditional probability tables based on parameter estimates from empirical species–stressor models), not qualitative data obtained from literature reviews or qualitative review methods.

3.4.2. Examples of applications

The use of BBN in terrestrial and aquatic CEAs has been increasing. As the number of stressors within ecosystems increases, so does the complexity of assessing and measuring additive or interactive effects, leading to incomplete information and the required data to support decision making by resource managers. We provide a summary of two examples that use BBN models within a CEA for terrestrial species. The first example uses a BBN model to assess the CE of recreational activities on brown bears (*Ursus arctos*) in Alaska (Fortin et al. 2016). The authors combined information from a literature review and a survey of brown bear experts into a BBN model to assist managers when evaluating the potential impacts of human recreational activities (e.g., camping, fishing, trail use) on brown bears (Fig. 5). Recreational activities were combined in summary parent nodes based on whether the activities were regulated or unregulated and if they occurred in habitat containing concentrated (e.g., salmon or salt-marsh meadows) or dispersed (e.g., cow parsnip) food resources for bears. The mechanisms of impact (represented by intermediate child nodes) included displacement from high- and low-quality habitat, energetic costs, and nutritional intake, while response variables (represented by output child nodes) included reproduction, cub survival, and adult survival. The completed model was used to compare potential consequences on reproduction, cub survival, and adult survival of five management scenarios containing multiple recreation activities in brown bear habitat (see tables 3–5 in Fortin et al. 2016) and identify management actions (e.g., public education, closures, placement of campground, trail, and bear-viewing sites). The second example integrates both TK and SK information within a BBN model to assess key questions about the CE of human stressors associated with multiple sectors (oil and gas, oil sands, forestry, coal and uranium mining, agriculture, hydro-electric dams) on the ecosystem health of the Slave River Delta region in northern Canada using 41 bird, fish, and water quality indicators and nine environmental health indices (Mantyka-Pringle et al. 2017; see Fig. 6). The authors rated overall ecosystem health as lower than in the past, with SK indicators ranked as moderate and TK indicators ranked as poor and used their results to identify the need for (*i*) transboundary water co-management agreements, (*ii*) effective communication of

Fig. 5. Example of a Bayesian belief network (BBN) model for an effects-based CEA. Figure represents a conceptual model of a BBN of impacts of human recreation on brown bears in Alaska, USA. Blue summary parent nodes (independent) summarize recreational activities with similar impacts. Beige intermediate child nodes (dependent) represent mechanisms of impact (displacement from high- and low-quality habitat, energetic costs, and nutritional intake). Green output child nodes represent response or demographic variables (cub survival, reproduction, and adult survival). From Fortin et al. (2016), reproduced (free of copyright) from PLoS One, Vol. 11, ©2016 Public Library of Science (PLoS).



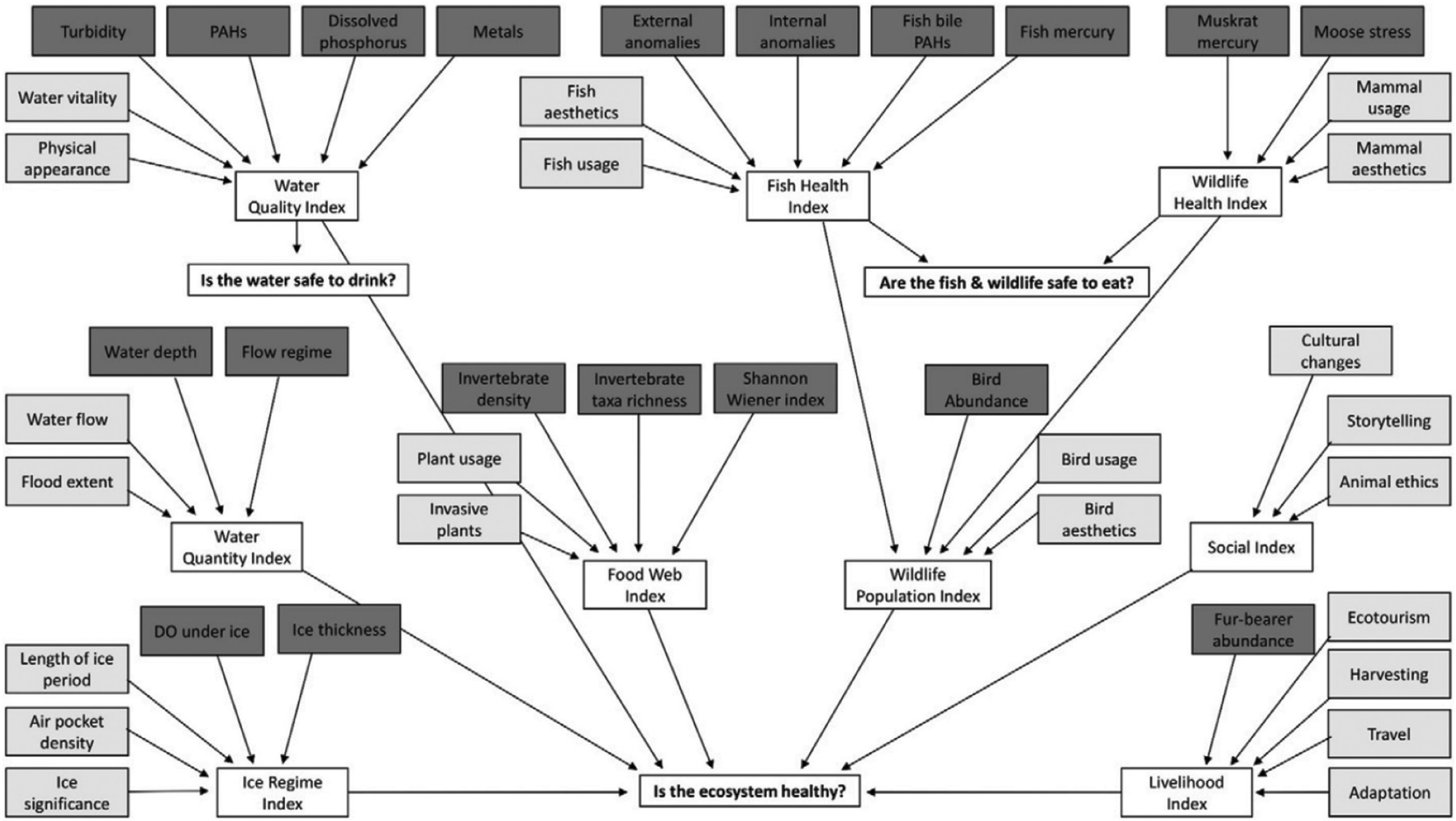


Fig. 6. Example of a qualitative review and Bayesian belief network (BBN) model. Figure represents a conceptual model of a BBN of the Slave River Delta, Northwest Territories, Canada using ecosystem health indicators and indices to address three questions: (1) Is the water safe to drink?; (2) Are the fish and wildlife safe to eat?; and (3) Is the ecosystem healthy? Coloured parent nodes (independent) represent 41 indicators where light grey nodes were informed by traditional knowledge (TK) and dark grey nodes were informed by scientific knowledge (SK) or data. White child nodes (dependent) represent response indices and output nodes/questions of interest (from Mantyka-Pringle et al. 2017, reproduced with permission of Environ. Int., Vol. 102, ©2017 Elsevier).

downstream effects that blends and synthesizes TK and SK, and (*iii*) global policy that supports indigenous people's cultures and rights. Additional examples from the published literature are listed in Table 3.

3.4.3. Advantages and limitations, CEA questions, and when/why to use

3.4.3.1. Advantages

A central advantage of BBN models is the ability to combine quantitative and qualitative data and associated data uncertainties when describing how stressors act to affect species or ecosystem components. This feature is valuable when empirical data are limited and when data, information, and knowledge from different disciplines and partner groups are available. Outcomes from multiple CEA projects using qualitative review (stressor rankings), habitat supply models (habitat rankings or suitability measures), or empirical species–stressor models (direction and size of effects from model coefficients; additive or interactive stressor combinations) can be incorporated into a BBN structure to address CEA questions. New data and information can quickly and easily be used to update or refine the model and conditional probability tables and formally validate the BBN predictions (e.g., comparing predictions from the BBN to new observations and calculating the overall accuracy). Sensitivity analysis can be used to determine how input variables influence outcomes and scenario analysis can be used to assess how alternate conditions and decisions influence outcomes. The use of Bayesian statistics to calculate probabilities and account for prior knowledge and missing data means that this approach is suitable for small data sets or data sets with missing data, which are typical in complex terrestrial systems with multiple species and stressors. Finally, the graphical format, similar to a conceptual model or influence diagram, depicts relationships among predictor (stressor) and response (outcome) variables resulting in an improved understanding of complex relationships and the analytical process or approach.

3.4.3.2. Limitations

Limitations of BBN methods include the following challenges: (*i*) technical expertise to develop the unique framework of a BBN and the time and resources to compile data/information to specify relationships and (*ii*) empirical data to specify the probability values associated with stressor effects within the conditional probability tables. If new data are required to populate conditional probability tables, partners should a priori develop rigorous sampling designs and survey protocols for terrestrial species to meet proposed project objectives.

3.4.3.3. Science-based CEA questions

See Table 4 to identify how empirical species–stressor model methods address key science-based CEA questions.

3.4.3.4. When/why to use – project-based and effects-based CEAs

"*When to use*": when data sources are a mixture of qualitative and quantitative sources and systems require complex assessments (e.g., multiple resource sectors and overlapping stressors, land use practices, and climate stressors), BBN models can provide a structured and flexible framework to identify and estimate effects of multiple stressors and investigate outcomes of specific management scenarios (see Steventon et al. 2006 and Fortin et al. 2016). "*Why to use*": despite data requirements (qualitative and quantitative data), and the technical expertise required to create the BBN model, this approach provides the opportunity for effective co-production of knowledge using TK and SK where each knowledge system is incorporated into the BBN (see Bélisle et al. 2018).

4. Recommendations and conclusions

We hope our review will help partners and decision makers understand the extent to which current analytical approaches (qualitative review, habitat supply models, empirical species–stressor models, and decision support models) can answer key science-based CEA questions. Although CEA has a long history, scientific review and guidance on analytical methods and outcomes has been surprisingly lacking, particularly for terrestrial species. Of the methods reviewed previously, empirical species–stressor models and decision support models that incorporate outcomes from species–stressor models are the only analytical methods that provide answers to most science-based CEA questions including understanding how different stressors combine to affect an individual species, identifying the significance of single and multiple stressor effects on populations, and quantifying the certainty of single and multiple stressor effects. These methods are consistent with a move toward larger, more complex, regional CEA within the peer-reviewed literature (Duinker and Greig 2006; Johnson et al. 2011; Noble et al. 2017) and the reports (CEAA Expert Panel Review 2017 of Environmental Assessment Processes), guidance (https://www.canada.ca/en/impact-assessment-agency.html), and acts (e.g., *Impact Assessment Act 2019*) issued by the Government of Canada.

Our review also highlights that although qualitative review and habitat supply approaches have been used extensively for project-based CEAs, inferences based on habitat change alone will not be sufficient to quantify and understand novel and complex multiple stressor effects that will be increasingly important to effective project-based or effects-based CEAs. We recognize that the higher data, time, financial, and technical requirements of empirical species–stressor models and BBN models may place these approaches beyond the scope of most project-based CEAs. Regional collaborations among partners (federal and provincial/territorial government agencies, conservation agencies, Indigenous communities and groups, academia, industry) to leverage data, expertise, and resources will be necessary to conduct rigorous effects-based CEA for terrestrial species at larger regional scales that address key science-based CEA questions. Subsequently, we provide suggestions and recommendations for moving forward given these constraints. First, we briefly touch on two analysis-related elements of CEA that deserve mention, but they were beyond the scope of this review.

First, our review was limited to analytical approaches based on western science and data (e.g., SK), and did not explicitly address inclusion of TK. Indigenous communities and groups may be uniquely and disproportionately affected by proposed developments and also possess long-term knowledge on species distributions, population changes, migration timing, and potential stressor effects that may be lacking or differ from conventional scientific data (e.g., Berkes et al. 2000; Parlee et al. 2012). How to best engage communities and include TK should be decided by Indigenous communities and groups. However, where of interest to stakeholders, we note that each of the approaches we review here offer options for co-production of TK and SK within the same analytical framework. Approaches already based on qualitative data (e.g., expert-based species–habitat models, fully qualitative reviews) may pose fewer technical challenges to including TK. For example, Polfus et al. (2014) and Bridger et al. (2017) used TK and local knowledge to develop species–habitat models for caribou and furbearers, respectively. Recent work has also demonstrated how TK may be blended with conventional quantitative modelling. For example, Johnson (2016) combined Elder knowledge of caribou harvest locations and river crossings with satellite collar point locations to quantitatively model caribou responses using habitat and disturbance variables. Mantyka-Pringle et al. (2017) used BBNs to blend TK, expert opinion, and scientific data to assess a suite of ecosystem health indictors in the Slave River Delta region of the Northwest Territories. Because many decision support frameworks, like BBNs, are specifically designed to combine quantitative and qualitative data types and sources, they are a frequent choice for studies incorporating TK into ecological models (Bélisle et al. 2018) and offer great potential for collaborative CEA.

Second, although we did not specifically review landscape simulation models, such models have and should continue to be a powerful tool for regional CEA and regional land use planning. Although simulation models require (*i*) technical expertise, (*ii*) extensive data to parameterize specific model components (e.g., vegetation dynamics, natural disturbance patterns, economic development), and (*iii*) accurate species–habitat or species–habitat–stressor models to include within the model or apply to model output, they allow users to track how a range of stressors (resource and land use, climate), natural disturbance regimes, and climate models under proposed or alternate development or management scenarios could affect VCs or terrestrial species. Several modelling platforms are available (e.g., ALCES: http://www.alces.ca; HexSim: http://www.hexsim.net; InVEST: http://www.naturalcapitalproject.org; LANDIS-II: http://www.landis-ii.org; NetLogo: http://ccl.northwestern.edu/netlogo; Repast Simphony: https://repast.github.io; SpaDES: http://spades.predictiveecology.org). Selection of a modelling platform will depend on the suitability of the platform to meet objectives (e.g., inclusion and quality of key economic process models to track changes in resource and land use stressors, climate models to track changes in climate, and vegetation models to track vegetation dynamics including natural succession and post-disturbance recovery) and the expertise of project members. Multiple examples exist in the literature including population changes of terrestrial species under: forest management scenarios (Shang et al. 2012; Mahon et al. 2014; Leston et al. 2020); forest management, industrial development, and infrastructure scenarios (Sutherland et al. 2016); forest management and climate scenarios (Wintle et al. 2005; Bonnot et al. 2013, 2017; Tremblay et al. 2018; Cadieux et al. 2020); and land use and infrastructure scenarios (Silva et al. 2010).

Conducting effective CEA has many challenges (Jones 2016; Hodgson et al. 2019) and appropriate use of analytical methods is no exception. We suggest the following recommendations to improve current analytical methods for both project-based and effects-based CEAs for terrestrial species.

1. Use standardized, consistent, and strategic implementation of CEA methods within smaller project-based CEAs, ensure results and data are available to better inform future assessments, and view these as essential building blocks on which a larger regional CEA (e.g., effects-based CEA) can be conducted using empirical species–stressor models and (or) BBN models. View CEA methods and results as hierarchical building blocks within and among CEA to improve efficiency, quality, engagement, and transparency. Use multiple methods within one project-based or effects-based CEA in a tiered or hierarchical framework. For example, use qualitative review to create conceptual models and describe the system and (or) the analysis process. Use specific habitat supply models (qualitative or quantitative) to assess changes in suitable habitat for a large number of terrestrial species. Use empirical species–stressor models and integrated population models to assess stressor effects for a smaller number of terrestrial species of high conservation concern where additional data from existing studies may be available. Use flexible decision support models like BBN models to incorporate qualitative and quantitative data, examine interactions among stressors, and evaluate management scenarios for a large suite of terrestrial species of stewardship, cultural, management, and conservation concern.
2. Prioritize the availability, accuracy, and consistency of data (spatial, nonspatial) for project-based and effects-based CEAs including sector, resource and land use stressor, climate stressor, landcover and habitat, and species data.
3. Support the continued use of decision support models (e.g., BBN) for project-based and effects-based CEAs that can represent complex ecosystems and species, combine multiple data sources, facilitate co-production of TK and SK, and report uncertainties.

Our review addresses a key challenge to project-based and effects-based CEAs: inconsistency in methods to conduct CEA (Hodgson et al. 2019). Providing a clear understanding of common CEA methods, science-based CEA questions, and when and why to use methods will allow proponents or partners to select appropriate designs and analyses before undertaking a project-based or effects-based CEA. Our review also identifies methods suitable for larger regional CEA projects, including Regional Assessments conducted for or by the IAAC. We suggest that effective regional CEA projects for terrestrial species will require a collaborative, multi-partner approach. Regional CEA projects provide the spatial scale necessary to capture the relevant range of stressor and stressor combinations encountered by terrestrial species, and, more importantly, the pooling of data, expertise, and resources necessary to conduct the analytical methods needed to answer key science-based CEA questions. We therefore encourage policy changes that support regional, effects-based CEA, and technical and analytical changes that support the production of rigorous, consistent, and transparent assessments of CE for project-based and effects-based CEAs in all jurisdictions in Canada. The latter requires (*i*) the implementation and integration of large-scale monitoring programs for terrestrial species (e.g., multi-taxa monitoring: Alberta Biodiversity Monitoring Institute; boreal bird monitoring: Boreal Optimized Sampling Strategy (Van Wilgenburg et al. 2020)); (*ii*) the consolidation and free access to stressor, landcover/habitat, and species data, and all data from past and current project-based CEAs; and (*iii*) the use of appropriate analytical methods to address science-based CEA objectives for terrestrial species.

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