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How well does LCA model land use impacts on biodiversity?—A comparison with approaches from ecology and conservation

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ABSTRACT

The modelling of land use impacts on biodiversity is considered a priority in Life Cycle Assessment (LCA). Many diverging approaches have been proposed in an expanding literature on the topic. The UNEP/SETAC Life Cycle Initiative is engaged in building consensus on a shared modelling framework to highlight best-practice and guide model application by practitioners. In this paper, we evaluated the performance of 31 models from both the LCA and the ecology/conservation literature (20 from LCA, 11 from non-LCA fields) according to a set of criteria reflecting: (i) *model completeness, (ii) biodiversity representation, (iii) impact pathway coverage, (iv) scientific quality* and (v) *stakeholder acceptance*. We show that LCA models tend to perform worse than those from ecology and conservation (although not significantly), implying room for improvement. We identify seven best-practice recommendations that can be implemented immediately to improve LCA models based on existing approaches in the literature. We further propose building a "consensus model" through weighted averaging of existing information, to complement future development. While our research focuses on conceptual model design, further *quantitative* comparison of promising models in shared case studies is an essential pre-requisite for future informed model choice.

KEYWORDS: Life cycle assessment; land use; biodiversity; UNEP/SETAC Life Cycle Initiative; ecological modelling

INTRODUCTION

The conservation of biodiversity represents a global priority due to its substantial contribution to human well-being[1] and ecosystem functioning[2]. The term 'biodiversity' is plural and encompasses a wide range of biological features with distinct attributes (ecological composition, function and structure), and nested into multiple levels of organization (genetic, species, population, community and ecosystem). Due in part to this complexity, the integration of biodiversity in decision-support tools, such as Life Cycle Assessment (LCA), remains a challenging issue[3,4]. LCA is a useful methodology to identify hotspots of environmental impact in supply chains, investigate the environmental performance of product design choices, and support the setting of environmental policies and regulations by public and private bodies.

Due to the importance of habitat loss and fragmentation in driving global biodiversity loss, accounting for biodiversity impacts resulting from land use practices remains a priority area of development in LCA[5]. In recent years, LCA researchers have drawn inspiration from the ecology and conservation literature when developing life cycle impact assessment (LCIA) models for biodiversity, including the use of meta-analyses[6,7] and Species–Area Relationship [8] (SAR) models of biodiversity loss[9,10]. More recently, LCIA developers have used species Habitat Suitability Models (HSMs) to derive land use indicators[11–14], used global conservation databases such as WWFs "Wildfinder" database[9,13,15,16] and adopted novel metrics reflecting non-compositional attributes of biodiversity (e.g. functional), such as indicators of critical ecosystem resources (e.g. dead wood[17]) and functional trait diversity[18].

Accompanying these recent developments is a need for critical reflection and guidance on best-practice approaches and directions for further research. The UNEP/SETAC Life Cycle Initiative (LC Initiative) has been working to identify indicators and models of biodiversity loss of particular promise for further application and development in LCA. The LC Initiative is a platform promoting life cycle thinking globally and engaging practitioners and stakeholders in order to achieve consensus on, and mainstreaming application of, life cycle tools and strategies. Under the efforts of its Phase 3 activities (2012-2017), it aims to enhance consensus and propose global guidance on existing environmental life cycle impact indicators, including those of biodiversity, used in the assessment of land use interventions[5].

The aim of this paper is to advance previous work of the task force[19,20] – as it applies to biodiversity – through a timely review of the literature on land use impact assessment (models, indicators). This furthers previous reviews of LCA methods[3,4] by taking a highly empirical approach (structured, standardized evaluation criteria and scoring) and opening up the scope of evaluation to a wide range of models from outside the LCA field (i.e. largely from the ecology and conservation literature). The paper is structured as follows. First, we present a refined conceptual framework of the major impact pathways leading from land use interventions ("inventory flows" in LCA) to final impacts on biodiversity ("endpoint" impacts in LCA). We describe the scope of our evaluation and the models identified for inclusion. We then describe a set of qualitative and quantitative criteria used for the evaluation and present the results, outlining differences in modelling approach and the strengths/weakness of the respective models. Based on these findings, we update the general modelling framework of Koellner *et al.*[20], identify best-practice guidelines from existing models, assess how well LCA models stand up to those from outside the field and provide recommendations on further development to fill conceptual gaps.

MATERIALS AND METHODS

Conceptual framework and conventions. We adopted a *conceptual framework* linking land use interventions to biodiversity impacts based on Koellner *et al.*[20] and Maia de Souza *et al.*[4], which represents the main drivers (not exhaustive) of biodiversity loss (Figure 1). Throughout the text we adopt the following conventions. We use the term biodiversity *indicator* synonymously with *metric* (e.g. species richness, abundance). We consider *models* to be collections of *approache*s or *sub-models* (e.g. the Species–Area Relationship) that model one or multiple impact pathways shown in Figure 1. In contrast, *method* is used to describe collections of models that address multiple impact categories within a larger assessment (e.g. Eco-Indicator 99 LCA method). We refer to “model performance” as an overall appraisal of how models integrate the various concerns reflected in our evaluation criteria (see below).

Literature review of biodiversity impact assessment models. Assessing how well LCA performs in quantifying potential biodiversity impacts resulting from land use requires a wide perspective that extends beyond the field of LCA[3,4]. We thus conducted a broad literature review of the LCA, ecology and conservation literature with the aim of identifying models compatible with the LCIA framework. Compatible models linearly attribute final impacts (e.g. proportion/number of species lost) to marginal inventory flows (land area converted and/or occupied, occupation time) of multiple land use types[19,20]. We adhered to the following broad inclusion criteria:

i. *Documentation*. The main description of the model was published in a peer-reviewed scientific journal. Unpublished (grey) literature was considered only if it contained important complementary information about the model, such as Louette *et al.*[21] in relation to Delbaere et al.[22]

ii. *Characterization*. Models were required to characterize biodiversity in at least two different classes or intensities of land use (e.g. forest or agriculture, intensive or extensive), *i.e.* that facilitate the goal of LCA to quantify the relative contribution of individual production processes (e.g. the use of land of a particular intensity) in driving overall biodiversity loss (e.g. all land use of all intensities).

To assist interpretation, throughout the text we differentiate between "LCA models", *i.e.* those specifically developed for LCA or already applied to LCA, and "non-LCA models", *i.e.* those not developed specifically for (or not yet applied to) LCA, largely originating from the ecology and conservation literature. This classification does not imply any *a priori* differentiation of model content or quality.

Model evaluation. We adopted the review framework used by the European Commission within the International Reference Life Cycle Data System on the evaluation of LCIA models and indicators[23]. We further included important methodological aspects important for biodiversity (e.g. representation of attributes and components, coverage of impact pathways, attention to scale) that were uncovered in recent review articles[3,4,19,20,24]. For each model, we first summarized the main model characteristics (Supporting Information (SI) Table S1), including the indicator(s) used to represent biodiversity, their position on the cause-effect chain (impact pathways depicted in Figure 1), as well as underlying data (e.g. literature data, expert opinion, etc.). We then evaluated each model using an explicit set of evaluation criteria.

We grouped sets of evaluation criteria under the following *evaluation categories*: (i) "Completeness of scope"; (ii) "Biodiversity representation"; (iii) "Impact pathway coverage"; (iv) "Scientific quality"; and (v) "Stakeholders acceptance". Under each of these evaluation categories, we compiled a set of criteria and sub-criteria, and qualitatively described the degree to which each model fulfilled each sub-criterion (SI, Tables S2–S6 with description of criteria in Table S7). For the first three of the evaluation categories, (i)–(iii), ordinal scores were attributed to each criterion using a three- or five-tiered, depending on the level of detail of the criterion. The scoring procedure is described in the SI (Table S7). Due to the ambiguous and subjective nature of categories (iv) and (v), we made only qualitative descriptions of model characteristics. A brief overview of the five evaluation categories follows.

i. *Completeness of scope.* This category reflects the overall scope of the model in terms of covering different land use/cover classes, degrees and type of land use intensity, geographic representation of the data and results, spatial resolution of the model and resulting "characterization factors" (CFs) and coverage of various taxonomic groups (SI, Table S2).

ii. *Biodiversity representation*. This category reflects the spatial-temporal representation of ecological attributes (composition, function and structure) at hierarchical levels of organization (genes, species, communities and ecosystems) of biodiversity. Additionally, it also recorded whether the index reflected heightened species extinction risk (e.g. by assessing effects on threatened species separately) to link to wider conservation assessments by, e.g., the IUCN (SI, Table S3).

iii. *Impact pathway coverage*. This category assesses the ability of the model to cover the relevant impact pathways shown in Figure 1 by evaluating their reflection in the indicators developed for the model (SI, Table S4).

iv. *Scientific quality.* We qualitatively considered overall scientific quality of the various models against three sets of criteria (SI, Table S5): scientific robustness (whether models reflect up-to-date knowledge, require frequent updates and are reproducible); uncertainty representation (consideration of uncertainties of indicators and model results); documentation and transparency (extent to which content is made available, including results and input data).

v. *Stakeholders acceptance*. This category assessed models from a stakeholder perspective of: LCA applicability (whether results can be interpreted as CFs for modelling occupation/transformation impacts in LCA); understandability of the models/indicators and the communication of uncertainties; and endorsement by an academic or governmental institution along with the neutrality of the model in balanced representation of economic sectors (SI, Table S6).

Model performance. For evaluation categories (i)–(iii), we assessed model performance as the geometric mean of the ordinal criteria scores within each category. These were used to rank models and compare LCA to non-LCA models. Because the data had variables data ranges (i.e. some criteria ranging 1–3, others 1–5), the geometric mean was appropriate as it is not biased by extreme values.

Modelling framework. When presenting results, we adhered to an adapted general framework for assessing land use biodiversity impacts suggested by Koellner *et al.*[20]. This framework is made up of three broad phases of 1) spatial model development, 2) data collection and approach, and 3) impact assessment (Figure 2). In the next sections, we follow this framework by first summarizing the review and describing the conceptual characteristics (model choices) and representation of evaluation categories (i)–(iii) (Figure 2.1 a–d). We then attempt to group the models based on qualitative characteristics (Figure 2.2 b). Finally, we discuss key components of model application and reflection of evaluation categories (iv) and (v) (Figure 2.3 a, c, d).

RESULTS AND DISCUSSION

Sampled studies. Our initial literature search identified 73 relevant published papers that fit the scope and topic of the review. Out of the 73, 31 publications matched the criteria for documentation and characterization and were included in the final assessment. Out of these 31 models, 20 were developed specifically for impact assessment in LCA and 11 originated from non-LCA fields (environmental policy, ecology and conservation). Table 1 presents a summarized overview of the models evaluated, listed by field. A detailed summary of each model, describing the main indicators of biodiversity, modelling approach, data sources and reference state, can be found in the SI (Table S1).

1) Spatial model development

Modelling choices

Indicator use. Species richness, abundance and functional diversity were the most common local indicator of biodiversity (used by 16, 8 and 6 model, respectively; Figure 3a). While richness only accounts for the change in the number of species locally[6,7,25,26], abundance-based models take into account population changes[11,27–29] and have been shown to be more sensitive to land use change[6]. Species extinction risk, habitat quality and composite indicators were less well represented (5, 4 and 2 models, respectively). Indicators of extinction risk explicitly translate changes in land cover and quality at the local scale into risks to regional or global losses of species [14,30,31]. Measures of habitat quality are largely subjective in nature, and include the “naturalness” of land cover classes (*i.e.* “Hemeroby” scores[32]) and the distance to artificial habitat edges[33]. Functional indicators included Human Appropriation of Net Primary Productivity (HANPP)[30,34–36], functional trait diversity[18], critical resources (e.g. dead wood[17]) and a range of ecosystem structural indicators summarized through meta-analysis[37]. Two models developed composite indices made-up of several weighted sub-indicators expressed in diverse units[17,38].

At the regional scale, indicators of the overall species pool size were most common (8 models), followed by habitat quality and extinction risk (7 models each) (Figure 3a). Species turnover rate in the regional species pool[25,26,39,40] and the absolute size of the pool[15] have been used as (part of) regional weighting factors for local impacts[15,26]. Alternatively, the effect of local land use on the size of the regional species pool has been modelled directly using the species–area relationship[9,10], the species–energy relationship[35,36] and habitat suitability models[11,12,41]. Regional habitat quality was generally represented by a quality-weighted sum of local values across the region[32,33,42], the relative change in ecosystem area[43–45]. Species extinction risk was expressed at various scales, from national changes in the number of threatened species[30] to global loss of species[14,31,46,47]. Other regional indicators included composite indices (3 models[15–17]), summed abundance/rarity values across a region (3 models[11,12,27,28,48]) and a single application of HANPP as a regional functional indicator[35,36]. In light of the above diversity of indicators, data availability is no longer a valid argument for *strictly* resorting to local species richness as a sole indicator. High-performing models should thus assess multiple facets of biodiversity, such as through composite indices, weighted-average, threat/rarity-weighted richness, etc. (Table 2).

Reference state(s). The reference state is relevant at both the local scale – as a benchmark habitat to standardize land use comparisons – and at the regional scale – as a baseline for calculating weighing factors (e.g. degree converted) or future scenarios of land use change. In order to improve clarity in future research, it would be beneficial to give these model components separate names, such as our suggested *local benchmark* and *regional baseline*, respectively. At the local level, we feel the term *benchmark* appropriately reflects the use of reference habitat in vegetation assessments as a desired and feasible target for ecosystem management, improvement or restoration (irrespective of whether this target is natural or human-modified)[49,50]. At the regional level, we feel the term *baseline* is adequate in capturing the concept of a starting point for scenario analysis (irrespective of whether this starting point is assumed to be a “natural” or current state of regional land cover). Thus, in the following text, we adopt the terms “local benchmark” and “regional baseline” throughout to clearly differentiate these two reference states at different scales, and use the term “reference” only when discussing aspects that apply to both scales (local and regional).

At the local scale, potential natural vegetation (PNV) was (often implicitly) assumed to be the local benchmark by 21 models (Figure 3b). 12 models investigated a current (e.g. average LU indicator value[40]) or human-modified benchmark (e.g. pasture in New Zealand[51]). In the non-LCA literature (i.e. ecology and conservation), a local PNV benchmark was implicit in almost all models considered, commonly referred to as “undisturbed”, “old-growth” or “primary” vegetation[27,29,33,35–37,48]. At the regional scale, both a PNV and current baseline were equally represented (12 models). Models using regional weighing factors and SAR-based approaches implicitly used a PNV baseline—e.g. ecoregion vulnerability/rarity[15–17,43], regional extinction models[9,10] or summed hemeroby/abundance/HANPP/habitat quality values[27,28,32,33,48]. These models estimate *average impacts* of production, incorporating historical loss into the assessment of impacts. In contrast, a current regional baseline was applied to scenario models to quantify the impact of future production (i.e. marginal impacts) over a set time horizon[9,11,12,33,41,44], and also for weighing factors based on current patterns of land use and intensity[39,40,42,45]. Models using species extinction risk as a regional indicator weight implicitly apply a current regional baseline due to the use of contemporary threat/rarity data[14,30,31,46]. Importantly, it should be recognized that a current regional baseline does not penalize historical land use change, only future marginal impacts. A current regional baseline may thus be more suited to consequential LCA, whereas a PNV regional baseline (with its focus on average impacts) is better suited to attributional LCA[11].

The usefulness of the PNV concept and the methodological challenges of its operationalization have been hotly debated in the ecology literature in recent years[52–55]. Some authors have interpreted PNV as the pre-historic, original, climax vegetation of a region[52,56]. This gives rise to difficulties in modelling PNV in areas with a long history of human modification due to past extinctions (e.g. of large mammals), human management history (e.g. fires, coppicing, harvesting), biological invasions (e.g. cultivated, naturalized and invasive species), soil changes (e.g. compaction, removal) and dynamic environmental change[52,54,56]. Interpreted as such, in regions where novel flora and fauna associations have co-evolved with human into ecosystems of high conservation value (e.g. European farmland biodiversity), the PNV reference (local or regional) could be counter-intuitive for conservation planning. Proponents of the PNV concept highlight the *hypothetical* and *future-orientated* nature of the original definition[57], which identified PNV as the "*hypothetical natural status of vegetation*" that could be identified "*by taking away human impact on vegetation*" and assuming succession is instantaneous[53–55]. This definition highlights the coarse-scale nature of PNV as a *hypothetical biotic potential* of a region based on patterns in existing remnants of maximally-undisturbed vegetation[53]. This definition is somewhat compatible with the concept of environmental change and anthropogenic influence, as it assumes that existing remnant vegetation must act as a source for colonizing species following abandonment and succession[57]. Further assumptions of this definition of PNV are that ecological communities and ecosystems are better adapted to less-disturbed environments, are more resilient within their natural range of variation and have an intrinsic value derived from their "naturalness" (i.e. autonomy from human influence)[49,58].

The latter definition appears to be closest to the use of PNV in impact assessment studies[50] (LCA and non-LCA), and we recommend future studies should make this explicit (Table 2). However, more theoretical and empirical research is needed on this issue, particularly investigating different behaviours of indicators as they approach the reference (e.g. monotonic increase, decrease or hump-shape[59]) and how reference choice affects assessment results. Several LCA studies have used alternative references[9,14,18,39,40,51], but sometimes lacking consistency and a clear theoretical justification (i.e. as it differs from PNV as a desired benchmark of biotic potential). For example, de Baan et al.[14] apply both a PNV and current local benchmark, but in both cases species richness is weighed by contemporary regional baseline data (i.e. *current* patterns in species rarity and threat). Michelsen et al.[51] also use both a PNV and current local benchmark, but PNV is used in both cases as the regional baseline for calculating regional weighing factors.

Finally, there are socio-economic and political implications of reference choice that require investigation. For example, by assuming pre-existing land use patterns, a current regional baseline ignores the historic legacy of land use change. This attributes no impact to current land use (i.e. land occupation), only to deviations from the current baseline (i.e. future transformation and occupation). LCA is a methodology designed to weight the ecological consequences of land use across different world regions that have developed their landscapes at different rates over the past decades, centuries and millennia[60,61]. A PNV baseline is far from conceptually sound, but offers a uniform layer of biotic potential from which to compare the effects of current and future land use. In contrast, a current baseline cannot differentiate responsibility of countries in driving global land use impacts and may influence the future distribution of socio-economic benefits of land use (e.g. through influencing environmental policy and product sourcing decisions). These implicit value choices are often neglected by model developers. More interdisciplinary research is required to better understand these important dimensions (Table 2).

Evaluation categories (i)–(iii) and performance of LCA vs. non-LCA models

We provide a detailed account of how each model meets our ecological evaluation criteria in the Supplementary Information (SI, Tables S2–S4 and S8). In the following section, we thus report only broad findings from evaluation categories (i)–(iii). In terms of *completeness of scope*, the spatial scale of assessment exhibited more pixel-based and sub-national/ecoregion approaches (8 and 10 models, respectively) than either national scale (5 models) or realm/biome/continent scale (7 models) (Figure 3c and SI Table S2). Coverage of biogeographic realms[62] biased the Palearctic (PA) realm (22 models), with generally equal representation of others (9–11 models). Taxonomic coverage varied from one (mainly plants) to 10 groups, while land use characterization ranged from 4 to 80 classes (SI Table S2). *Biodiversity representation* showed a majority of models used indicators at the community (26) and ecosystem (21) level, although several models (13) assessed species-level effects (i.e. average population or range size change) or used indicators of conservation risk (14 models). Compositional indicators were most common (27 models), followed by function (9 models) and structure (6 models; SI, Table S3). Genetic components were only assessed as part of a meta-analysis in a single study[37]. *Impact pathway coverage* was most common for direct, local degradation and conversion of habitats (29 models; SI, Table S4). Wider landscape effects were often assessed in relation to the area of remaining habitat, indirectly reflecting habitat patch size (16 models), but largely omitting the effects of other fragmentation effects, namely patch isolation (2 models) and edge effects (3 models). Otherwise invasive species and below-ground impacts were similarly represented by 5 and 8 models, respectively. While only Gardi *et al.*[38] explicitly targeted below-ground diversity, we included meta-analyses as indirectly assessing this impact pathway through including datasets on soil arthropods[6,37].

In general, LCA models did not perform significantly worse than those from non-LCA fields (overlapping 95% confidence intervals in Figure 3e). However, there was a trend towards better model performance from non-LCA fields (higher mean score values), implying some potential for improvement. However, the ranking of individual models showed varying performance across categories (Figure 3f). To identify best-practice guidelines, we list the top five ranking LCA models in each category: *completeness of scope*[6,10,17,18,42], *biodiversity representation*[14,15,30,42,44] and *impact pathway coverage*[9–11,16,45] (based on rankings in Figure 3f). To help combine desirable properties of existing LCA models with novel features of non-LCA fields, we provide some broad recommendations and specific operationalization notes for future research in Table 2. At the same time, we urge caution in interpreting the individual rankings (Figure 3f) as definitive measures of model performance, as they lack qualitative consideration and do not reflect categories (iv) and (v). Additionally, the geometric mean assumes linear scaling of ranked scores (i.e. the performance increase between scores 1 and 2 is comparable to between 4 and 5), which may not be the case.

2) Data collection and modelling approach

Top-down, bottom-up and expert-based

In assessing the literature on the biodiversity effects of land use, Newbold *et al.*[29] classify approaches as based on SAR, Species Distribution Models (SDMs) or meta-analyses of the literature. Based on our review, we can expand this list to include the Species–Energy

Relationship (SER; linking HANPP to species loss), HSMs (*i.e.* measure habitat associations of species rather than their environmental/climatic niches as in SDMs), expert judgement (e.g. of "naturalness" or species responses), regression analysis (e.g. of land use composition and threatened species density), composite indices of diversity (*i.e.* composed of several sub-indicators, often combined with expert judgement in setting weights) and simple area metrics (e.g. rare ecosystem area, not based on the SAR) (SI, Table S1). To simplify, we classify approaches as (a) *top-down, process-driven*, (b) *bottom-up, pattern-driven* and (c) *subjective, judgement-driven*, and describe each in turn. These approaches are not mutually exclusive within a single model, and are frequently used in combination (e.g. meta-analysis used to derive land use sensitivity scores in an adapted SAR model[9]).

Top-down, process-driven. *Top-down* approaches constitute fitting a parametric function to diversity data based on a pre-defined mechanistic relationship describing an observed process. The SAR is particularly common[9,25,26,39–41,63], but recent work has translated HANPP[35,36] into species losses based on the SER, which is theoretically grounded in the "metabolic theory of ecology"[64]. Unharvested NPP has been proposed and operationalized as an indicator of land use impacts to life-support functions in LCA[43,65,66], and a complementary ecosystem exergy approach has been proposed[67] that is compatible with the SER. The HANPP approach is promising in that it allows the use of high-resolution, global, standardized and continuous biomass use intensity data[68]. However, the HANPP–species diversity relationship has only been parametrized in Austria[35,36,69] and is currently insensitive to spatial context or non-equilibrium conditions (limiting its usefulness of predicting species/ecosystem impacts of conservation relevance). At the same time, the SAR represents a traditional approach in ecological research. Limitations have included the inability to account for different grades of land use intensity and non-equilibrium conditions[4]. Recent work has produced adapted SARs that account for the influence of habitat fragmentation, diverse land uses and edge effects[70–73], which have been adapted for land use impact assessment in LCA studies[9,10].

Another approach that we consider top-down is the direct use of habitat area and quality metrics as an indicator of biodiversity. This is equivalent to fitting SAR with a straight-line function describing the habitat area/quality–biodiversity relationship. Examples include one application of the InVEST tool[33], the regional weighting step of the "naturalness"/hemeroby approach[32], the country-level aggregates of the "Biodiversity Intactness Index" and "Mean Species Abundance"[27,28,48], the area component of rare/unique ecosystem and Biotope models[43–45]. In such cases, habitat quality (e.g. "naturalness"[32]) or ecosystem characterization (e.g. identifying a "critical" biotope[44]) is often determined through other modelling approaches (described below), such as expert judgement or the analysis of empirical data on important biodiversity elements.

Bottom-up, pattern-driven. We characterize *bottom-up* approaches as based on extracting statistical relationships from different types of empirical data at various scales (e.g. meta-analysis of comparative land use studies). In our review, meta-analyses were common, and covered indicators of species abundance/probability of occurrence[28,29,48], richness[6,7,15,25,26,40], functional diversity[18], or combinations of the above[18,37]. Meta-analyses characterize predefined land use classes according to relationships in plot-level data that generally range up to several hectares in scale (e.g. vegetation plots, bird surveys) for each land use class/treatment (i.e. characterize local impacts). Other bottom-up approaches included aggregating species-level data in the form of species HSMs, transformed into spatial models when applied within a species' range map (*i.e.* identifying the suitability of land cover types within a species' known range using HSMs[74]). The resulting spatial HSMs were then combined in various ways to construct aggregate indicators for specific land use classes or assess impacts across land use change scenarios. Again, approaches may be combined, and subjective judgement as well as syntheses of local studies may be used to identify habitat associations used in HSMs. Resulting HSM-based indicators in our review include the regional persistence or abundance of focal species[21,41] and rarity, threat and/or evenness-weighted richness[11,14]. An identical approach could be pursued using SDMs, although our review did not uncover any specific examples.

A final bottom-up approach consisted of linking threatened species occurrence (either nationally or globally) with surrounding land cover composition[14,30,31] or economic sector composition[46] to derive statistical estimates of threat causation. The approach of Lenzen *et al.*[31,46] uses larger scale (national or global) data, and is thus restricted to coarse-grained analysis. In contrast, the approach of Matsuda *et al.*[30] uses pixel level information on species range size, population density and decline rates to produce spatially differentiated models (at 10 km resolution for Japan). The model of de Baan *et al.*[14] combined species HSMs with weights for extinction threat and rarity (rescaled IUCN threat status and inverse extent of occurrence) to unite local richness values with global indicators of extinction risk. This method produces high-resolution, globally-relevant characterization factors (at 900 m resolution), but is restricted in geographic (East Africa) and taxonomic (mammals) coverage, thus limiting its usefulness for global scale analyses (e.g. international supply chains).

Subjective, judgement-driven. The final class of approaches we discuss are those that operate with a significant input from "experts", classed as *subjective* approaches. These include more abstract, weighted composite indicators that are context-specific[38], or use qualitative descriptive criteria in determining "naturalness"[32]. Other examples include the use of composite indices to structure the combination of empirically-based sub-indicators, such as the WWF ecoregion "Conservation Risk Index"[15], regional "Vulnerability Index"[16], or the "Conditions for Maintained Biodiversity" index[17]. In addition, while empirical data (e.g. endemic species, habitat area) is often used to classify the importance of ecosystems in the rare/unique ecosystem and "Biotope" approaches[43–45], there is a significant degree of subjective choice as to cut-off points used in such classifications (and thus a multitude of possible options). A further common use of subjective judgement is through interviews with experts to derive predictions for changes in well-defined biodiversity variables (e.g. species abundances, richness change) where empirical data is lacking or inadequate[21,27,42].

3. Impact assessment

Key components. Based on our revision of impact pathways (Figure 1), and how these are translated into modelling choices (SI, Tables S1 and S4), there is a recognized need to model characterization factors in terms of both (i) local damage factor for land use linked to the functional unit, and (ii) regional "state and pressure" weight to reflect broader biodiversity patterns and processes surrounding the location of land use[75]. Following Koellner et al.[20], we recommend the use of the terms local and regional "Biodiversity Damage Potential" (BDP_L and BDP_R). The preceding discussion on indicator choice, reference state, modelling approach etc. should make it clear that a large variety of solutions have been found in quantifying BDP_L and BDP_R.

Scientific quality and uncertainty. Within our review, almost all models calculated BDP_L and BDP_R using data from the past two decades. Almost half of the models relied to some degree on local monitoring data, although only a few models[28,29,48,65,66] are directly linked to ongoing monitoring efforts and (assumed) data updates. Primary data underpinning models was generally only partially presented/available, *i.e.* covering only some of the indicators[63,76], or having monitoring data only for a specific case study[28,46,48]. Consequently, model updates are possible, but new data must be gathered. Approximately half of the models included some type of statistical analysis of indicator uncertainty resulting from modelling (e.g. standard error, standard deviation, sensitivity analysis) (SI Table S5). However, it was difficult to assess the completeness of the uncertainty analysis (e.g. for the studies carrying out sensitivity analysis[15,31,47], the parameters that were included, their simulated range, etc. were often not clearly described). Only eight studies included both indicator and *model* uncertainty (i.e. exploring scenarios and considering the uncertainty of the model itself). Specifically, 12 models included scenario modelling (I.e. model uncertainty) and 10 considered only the uncertainty of the indicator values (SI, Table S5). All studies including both types of considerations provided also uncertainty figures for the indicators.

The majority of evaluated models are well documented (SI Table S5), although extended descriptive versions of some of the models are not available in English (e.g. in German[45] and Japanese[65]). Regarding the usability of results, LCA models often present calculated CFs, with application in case studies. Input data is either transparently reported (e.g. as online database) or reported just as summary data in the published study. Regarding the possibility to reproduce the study and/or calculate new CFs, several models explicitly mention their underpinning indicators and parameters[14,15,18,28,37,48]. In some cases, models tested the effect of using alternative parameters[9,18] or had different modelling choices proposed to the practitioner[63,76].

Stakeholder acceptance. Amongst the models developed in the non-LCA domain, more than half (7) have the potential to be compatible with LCA and potentially implemented in software[21,31,37,38,46,48]. The major limitations to LCA compatibility are the presentation of aggregated results[33] and the requirement of case-specific data[44,77], which limits the level of possible implementation in software (SI Table S6). Most of the local indicator results are presented as relative values, either as a ratio/percent (i.e. 0-1; 0-100%), or expressed relative to area affected (m^2.y PDF). Most of the models evaluated haven't been endorsed by any authoritative governmental or academic body, with some exceptions[21,28,32,48]. In addition, in some methodological developments[45,65,66], local or national scientific bodies have been involved (international academic body endorsement). Another point to take into account is sectoral representation of the model, which means capability of the model to cover all different land use classes from different and relevant geographic origins and different areas and sectors (urban, industry, forestry, agriculture, etc.). Unfortunately most of the models do not provide CFs for all sectors. Agriculture and forestry are the activities within the primary sector most commonly addressed (SI Table S5). However, different intensities of production are only partially included[76]—mainly when they are models specific for agricultural purpose—or not included in any way due to a large scope of assessment[9]. In contrast, some models are more detailed in land use classes but they only provide information for specific regions, usually Europe or North America[18,21,25,26,32,38,40].

Toward consensus-building

The purpose of this concluding section is to discuss possibilities of finding consensus among this diversity of models. Specifically, by taking the best that each model has to offer (from both within and outside of LCA) we might avoid the standard recommendations of future development of a presumed “perfect model” (a potentially Sisyphean task). Major problems in combining existing models are the variety of land use classification typologies, taxonomic groups assessed, indicators and reference states etc. One pragmatic way of building consensus could be to construct a weighted average of available indicators from the literature for both BDP_L and BDP_R. We have attempted to provide a flow diagram of the main steps of such a procedure in Figure 4. While this will be a challenging task, we outline what we see as the key steps, and recommend at least some future research effort is invested in consolidating the results of the enormous efforts that have already been expended in the literature (which by no means discourages the simultaneous development of new models and approaches). Of particular importance in such a process is the weighing of different indicators across models when calculating a weighted average. This could be based on literature review (e.g. of essential biodiversity variables[78]), expert interviews of what ecologists think are the “right” indicators of biodiversity change (e.g. in terms of precision, bias and accuracy), or based on a structured evaluation such as the criteria and scores developed in this review (SI, Tables S2–S4 and S8).

While our review focused on conceptual issues of model development, we did not investigate the actual empirical differences of applying different models to the same case studies. Until this type of work is conducted, it remains difficult to predict how differences in modelling approach translate to different impact assessment results. This comparative focus is critical, and has been only sparsely investigated in the literature[11,12,14,43,51]. Achieving both of these aims of model combination and empirical comparison would represent an important step in the context of the UNEP/SETAC global consensus building effort.

ASSOCIATED CONTENT

Detailed information is provided on models evaluated as well as list of evaluation criteria along with quantitative score. This material is available free of charge on the Internet at http://pubs.acs.org.

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FIGURE LEGENDS

Figure 1. Conceptual model of impact pathways for land use impacts on biodiversity, moving from land interventions (occupation and transformation) to resulting environmental pressures and impacts at the midpoint and endpoint level (within the "Area of Protection" of Ecosystem Quality). Adapted from Koellner *et al.*[20] and Maia de Souza *et al.*[4].

Figure 2. Generalized modelling framework for assessing biodiversity impacts in LCA. Adapted from Koellner *et al*.[20] The grey boxes (2a "Generic considerations" and 3b "Modelling scope") are not discussed in detail in this paper. LUC = Land Use Change, SAR = Species Area Relationship, SER = Species Energy Relationship, HSMs = Habitat Suitability Models, BDP = Biodiversity Damage Potential, FU = Functional Unit

Figure 3. Summary data for the 31 models: a) the type of local and regional biodiversity indicators included in models; b) the assumed reference state at local and regional scales; c) the spatial resolution of assessment; and d) the coverage of biogeographic realms; e) geometric average model performance according to field (LCA, non-LCA) for evaluation categories (i)–(iii) (Scope, Representation, Coverage); f) Geometric mean scores per evaluation category and model. Models can contribute multiple values to graphs a–e (e.g. the use of both local and regional indicators, or analysis of multiple reference states), thus the sum of values does not equal number of models evaluated. PNV = Potential Natural Vegetation

Figure 4. Flow diagram of a proposed approach for building a consensus land use model through weighted averaging of existing model data and results. Example indicators and data sources are presented for illustration purposes only (Koellner et al.[19]). LC(C) = Land Cover (Change)

TABLES

Table 1. Models included in the evaluation listed by research field (LCA or non-LCA) with a short summary of their features. If multiple publications were used to evaluate a single model, we list the main peer-reviewed publication first (i.e. that used as the basis for evaluation) followed by supplementary grey or published literature. In the main text and figures we refer only to the first cited article in this list. * = Two applications of the InVEST model[79] were treated separately due to different approaches, geographic context and data used.

Model reference	Comments
	LCA models
Brentrup *et al*. (2002)[32]	The "Hemeroby" model (or "degree of naturalness") of land cover types, which represents a ranking of land cover types based on their intensity, converted to a cardinal scale.
Burke *et al.* (2008), Kyläkorpi *et al*. (2005)[44,77]	The "Biotope" model of the area change of ecological mapping units ("Biotopes") wihtin before/after land use scenarios.
Chaudhary et al. (2015)[10]	Ecoregion-based SAR approach with a species Vulnerability Score as regional weight.
Coelho and Michelsen (2014), Michelsen et al. (2014)[16,51]	"Conditions for maintained biodiversity" (CMB) based on local hemeroby at a regional Vulnerability Score.
de Baan *et al.* (2013a)[6]	Meta-analysis approach for CFs based on species richness indicator, regionalized for global biomes.
de Baan *et al*. (2013b)[9]	Ecoregion-based SAR approach with both reversible and permanent impact categories.
de Baan *et al.* (2015)[14]	Spatially-explicit using mammal habitat suitability models to derive indicator of threat and rarity weighted species richness.
De Schryver *et al*. (2010), Goedkoop *et al*. (2009)[63,76]	ReCiPe 2008 LCA method, land use sub-model, based on species richness indicator for British plants.
Elshout *et al*. (2014)[7]	Meta-analysis approach for CFs based on species richness indicator for plants.

Geyer *et al*. (2010a, 2010b)[11,12]	Spatially-explicit, scenario model using mammal habitat suitability models and multiple indicators (richness, rarity, evenness, hemeroby).
Jeanneret *et al*. (2014, 2006)[42,80]	Central-European agricultural LCA model (SALCA-biodiversity) based on expert judgement of abundance change with land use/management.
Schmidt (2008), Koellner (2000)[25,26]	Species-Pool Effect Potentials (SPEP) based on local species richness and SAR-based regional analysis.
Koellner and Scholz (2007, 2008)[39,40]	Central-European model of "Ecosystem Damage Potentials" (EDP^S) for plant species richness across CORINE land cover classes.
Lindeijer (2000)[34]	Early method using vascular plant species diversity and Net Primary Productivity (NPP) as local indicator.
Matsuda *et al*. (2003), Itsubo and Inaba (2012), Li *et al.* (2008)[30,65,66]	Indicator of "Expected Increase in Number of Extinct Species" (EINES) and NPP appropriation caused by land use.
Michelsen (2008)[17]	"Conditions for Maintained Biodiversity" model, based on key biodiversity indicators for Scandinavian forestry.
Mueller *et al.* (2014)[15]	Ecoregion-based model of local species richness of plants and a regional "species pool", "irreplaceability" and "vulnerability" weight.
Souza *et al*. (2013)[18]	Functional diversity model based on meta-analysis of species composition and functional trait data across land cover types.
Urban *et al*. (2012)[45]	Area-based method using loss of habitat "valuable to biodiversity" and a regional weight (land cover diversity, fertilizer use).
Vogtländer *et al.* (2004)[43]	Rare ecosystem model based area changes of prior identified high conservation value ecosystems.
	Non-LCA models (ecology and conservation)
Alkemade *et al*. (2009, 2013)[28,48]	GLOBIO3 model using indicator of "Mean Species Abundance" of "original species" for impacts of land use and fragmentation.
Gardi *et al*. (2013)[38]	"Soil Pressure Index" based on models of the intensity of 7 pressures on soil biodiversity.
Gibson *et al*. (2011)[37]	Meta-analysis of multiple indicators of tropical forest biodiversity at the local level in response to land use and land use change.
Haberl *et al*. (2004, 2005)[35,36]	"Human Appropriation of Primary Productivity" (HANPP) model for assessing land use impacts via a Species Energy Model (SEM).
Leh *et al*. (2013)*[33]	Application of the InVEST model to future scenarios of biodiversity change in West Africa, using a habitat quality indicator.

Lenzen *et al.* (2009, 2013)[31,47]	Species threat indicator (Red List Index) regressed at the national scale against land cover patterns.
Lenzen *et al*. (2012)[46]	Species threat indicator linked to Input-Output inventory items based on the IUCN Red List on threat causes, aggregated nationally.
Louette *et al*. (2010), Delbaere *et al*. (2009), Overmars *et al*. (2014)[21,22,81]	"BioScore" model of biodiversity change, based on literature and expert judgement of focal species sensitive to land use intensity.
Nelson *et al*. (2009)*[41]	Application of the InVEST model to future scenarios of biodiversity change in the USA (OR), using a mammal habitat suitability models.
Newbold *et al.* (2014)[29]	Global meta-analysis of primary data on local (rarity-weighted) species richness and abundance change due to land use.
Scholes and Biggs (2005)[27]	The "Biodiversity Intactness Index", based on expert judgement of population abundance change across land cover classes.

Table 2. Best-practice recommendations for LCA model development (in order of priority). For all recommendations, some implementation notes or examples are provided based on our review.

Recommendation	Rationale	Implementation notes
1) Take a multi-dimensional approach in indicator development	The multidimensional nature of biodiversity requires plural indicators spanning attributes (function, structure, composition), taxonomic groups and spatial scales. The data availability argument for adopting species richness no longer applies given numerous alternatives.	Meta-analysis provides a useful way to cover many aspects (composition, structure, function, abundance) of local biodiversity[18,37,48]. Several models complement richness with species weights (e.g. threat/rarity). Recent global datasets on evolutionary (genetic) distinctiveness[82,83] could be integrated into regional model components.
2) Develop and document both local and regional model components (BDP_L and BDP_R)	Both a local and regional components (BDP_L, BDP_R) are critical to modelling biodiversity change, and published models should present local/regional indicator results separately.	Widespread in many LCA models (e.g., through regional weighing factors[16], SAR-based approaches[10] or threat/rarity-weighted richness metrics[14]), but recent models[6,7] only assessed local impacts, which should in future be presented as a *component* of a complete model.
3) Reflect intrinsic value and vulnerability of biodiversity in the regional component	In order to differentiate regional importance for biodiversity, BDP_R should reflect both intrinsic values (e.g. rarity, endemism, irreplaceability) and vulnerability (e.g. the current state, threat status, future pressures).	Already exists in several ecoregion and species-based models (i.e. endemism, rarity, IUCN threat status, past conversion), but could be complemented with indicators of land use change (e.g. deforestation rates[84]), fragmentation (e.g. patch size effects[48]) and genetic indicators (e.g. evolutionary distinctiveness[82,83]).
4) Differentiate basic extensive/intensive land management practices	To help to differentiate between resource-intensive and extensive production systems, better differentiation of biodiversity change with land use intensity is required.	Several models assess intensiveness with at least 2 intensity classes for agriculture, forestry and built-land (e.g. GLOBIO3[48]). This may lead to trade-offs with other factors (i.e. limited regionalization or taxonomic differentiation). Yet without intensity information, decisions on land use options are highly limited.
5) Assess and report model and indicator uncertainty	Reporting of indicator and model uncertainty is a crucial component of decision-making, and should be a part of all models in some form or another.	Empirically-based, model uncertainty (i.e. across all model components and parameters) exists in several models (e.g. Monte-Carlo simulation[9]). Simulated uncertainty (e.g. sensitivity analysis, expert judgement[21]) is also helpful to document

		model robustness.
6) Interpret the PNV reference at local and regional scales as "*hypothetical biotic potential*"	The use of PNV has a strong precedence in the impact assessment literature, but has multiple interpretations. The definition as *hypothetical biotic potential* based on patterns in extant, minimally-disturbed vegetation (not "original" or future successional state) appears closest to its application in the impact assessment literature.	Already an implicit feature of the majority of models for local indicators, but more clarity is needed on the meaning of "biotic potential" regarding recent introductions/extinctions, non-plant taxa, active restoration etc., along with the expected behaviour of indicators around the reference value (e.g. monotonic increase, hump-shaped[59]).
7) Experiment with alternative local and regional reference states	The choice of reference state has ecological, socio-economic and political implications. More studies are needed to investigate the advantages and disadvantages of alternative states (e.g. current or human-influenced LC).	Several studies have applied a current reference, either locally (benchmark) or regionally (baseline)[9,41,51]. However, both theoretical and empirical work is required to understand the various implications (ecological or otherwise) of choosing a particular reference state at a particular scale.

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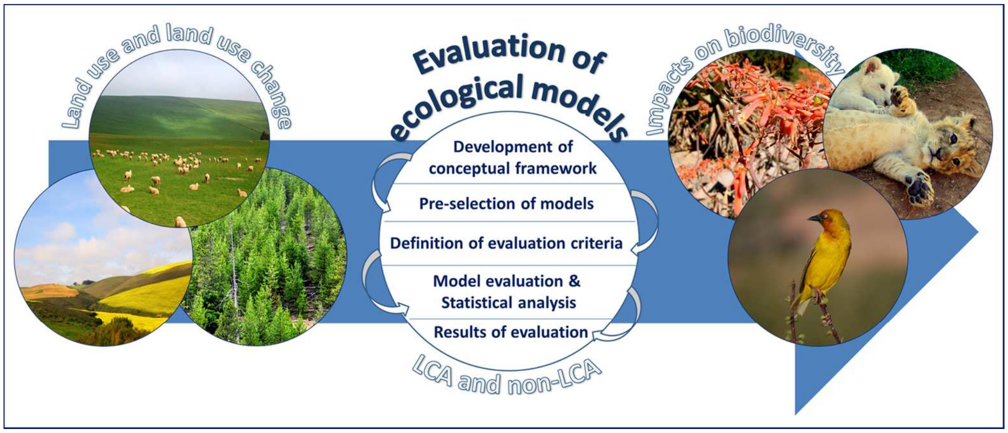

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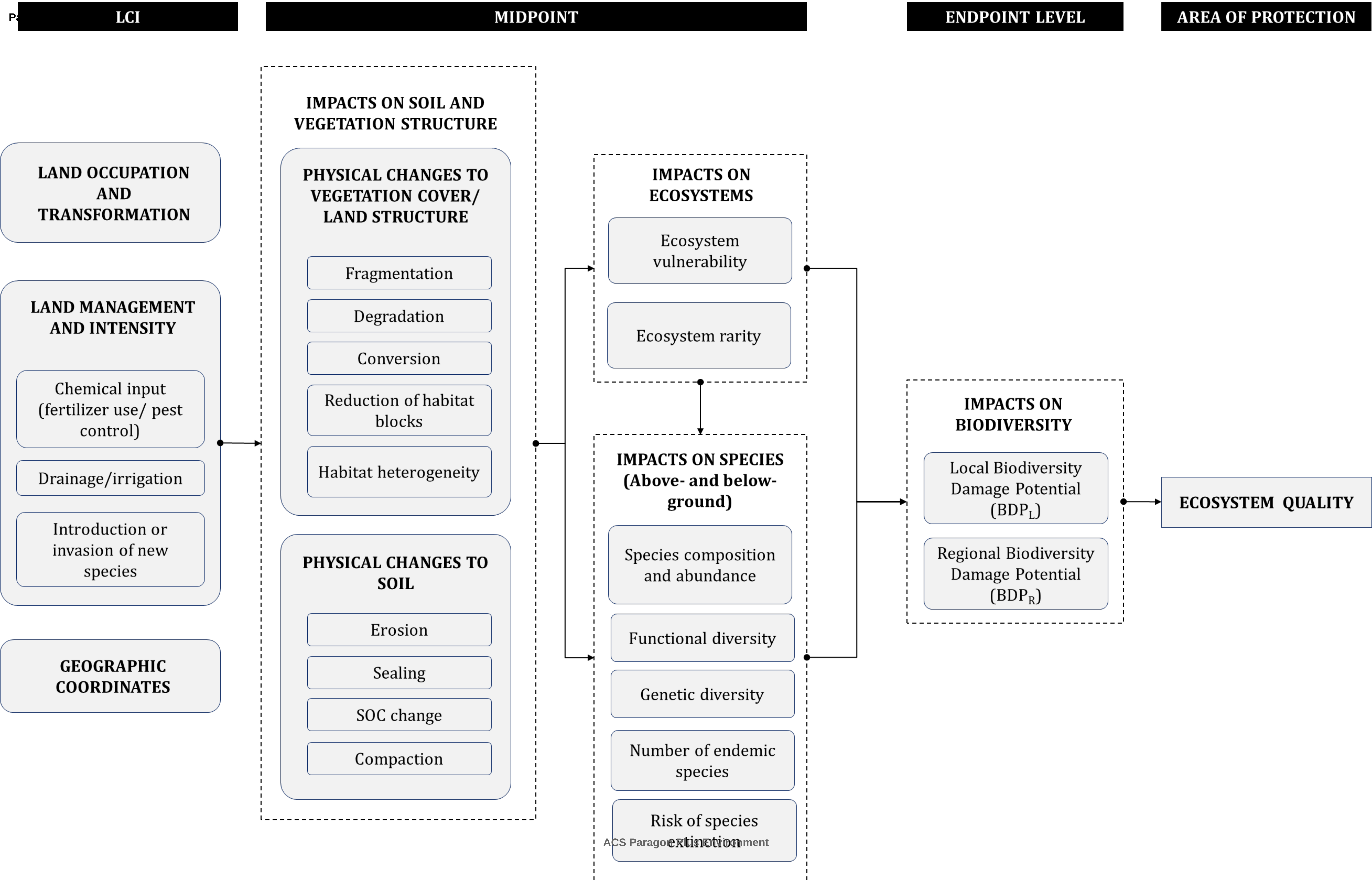
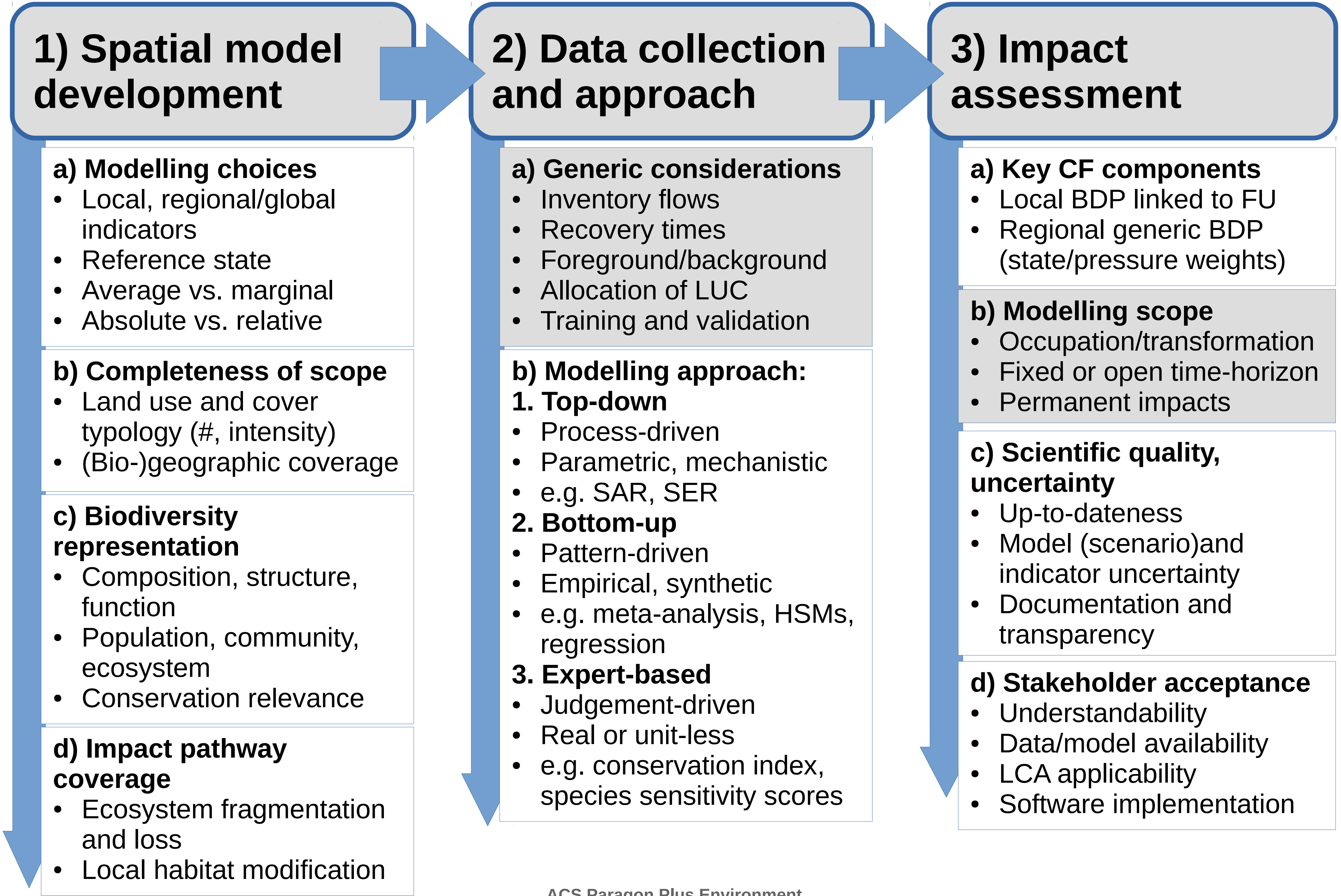
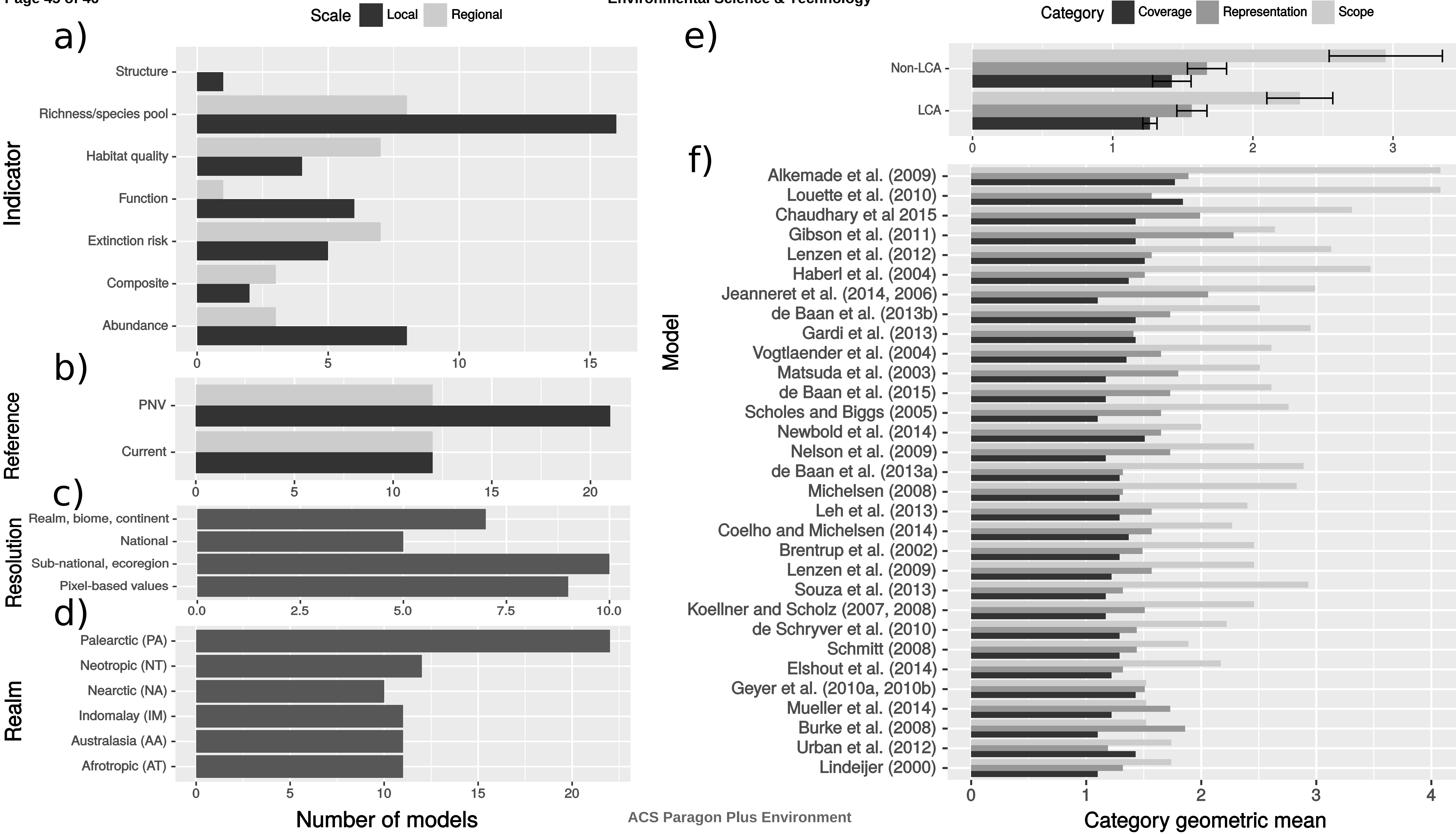

LCI
MIDPOINT
ENDPOINT LEVEL
AREA OF PROTECTION
LAND OCCUPATION AND TRANSFORMATION
LAND MANAGEMENT AND INTENSITY
Chemical input (fertilizer use/ pest control)
Drainage/irrigation
Introduction or invasion of new species
GEOGRAPHIC COORDINATES
IMPACTS ON SOIL AND VEGETATION STRUCTURE
PHYSICAL CHANGES TO VEGETATION COVER/ LAND STRUCTURE
Fragmentation
Degradation
Conversion
Reduction of habitat blocks
Habitat heterogeneity
PHYSICAL CHANGES TO SOIL
Erosion
Sealing
SOC change
Compaction
IMPACTS ON ECOSYSTEMS
Ecosystem vulnerability
Ecosystem rarity
IMPACTS ON SPECIES (Above- and below-ground)
Species composition and abundance
Functional diversity
Genetic diversity
Number of endemic species
Risk of species extinction
IMPACTS ON BIODIVERSITY
Local Biodiversity Damage Potential (BDP_L)
Regional Biodiversity Damage Potential (BDP_R)
ECOSYSTEM QUALITY


1) Spatial model development
a) Modelling choices
• Local, regional/global indicators
• Reference state
• Average vs. marginal
• Absolute vs. relative
b) Completeness of scope
• Land use and cover typology (#, intensity)
• (Bio-)geographic coverage
c) Biodiversity representation
• Composition, structure, function
• Population, community, ecosystem
• Conservation relevance
d) Impact pathway coverage
• Ecosystem fragmentation and loss
• Local habitat modification
2) Data collection and approach
a) Generic considerations
• Inventory flows
• Recovery times
• Foreground/background
• Allocation of LUC
• Training and validation
b) Modelling approach:
1. Top-down
• Process-driven
• Parametric, mechanistic
• e.g. SAR, SER
2. Bottom-up
• Pattern-driven
• Empirical, synthetic
• e.g. meta-analysis, HSMs, regression
3. Expert-based
• Judgement-driven
• Real or unit-less
• e.g. conservation index, species sensitivity scores
3) Impact assessment
a) Key CF components
• Local BDP linked to FU
• Regional generic BDP (state/pressure weights)
b) Modelling scope
• Occupation/transformation
• Fixed or open time-horizon
• Permanent impacts
c) Scientific quality, uncertainty
• Up-to-dateness
• Model (scenario)and indicator uncertainty
• Documentation and transparency
d) Stakeholder acceptance
• Understandability
• Data/model availability
• LCA applicability
• Software implementation


Scale
Local
Regional
a)
Indicator
Structure
Richness/species pool
Habitat quality
Function
Extinction risk
Composite
Abundance
b)
Reference
PNV
Current
c)
Resolution
Realm, biome, continent
National
Sub-national, ecoregion
Pixel-based values
d)
Realm
Palearctic (PA)
Neotropic (NT)
Nearctic (NA)
Indomalay (IM)
Australasia (AA)
Afrotropic (AT)
Number of models
Category
Coverage
Representation
Scope
e)
Non-LCA
LCA
f)
Model
Alkemade et al. (2009)
Louette et al. (2010)
Chaudhary et al 2015
Gibson et al. (2011)
Lenzen et al. (2012)
Haberl et al. (2004)
Jeanneret et al. (2014, 2006)
de Baan et al. (2013b)
Gardi et al. (2013)
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Matsuda et al. (2003)
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Scholes and Biggs (2005)
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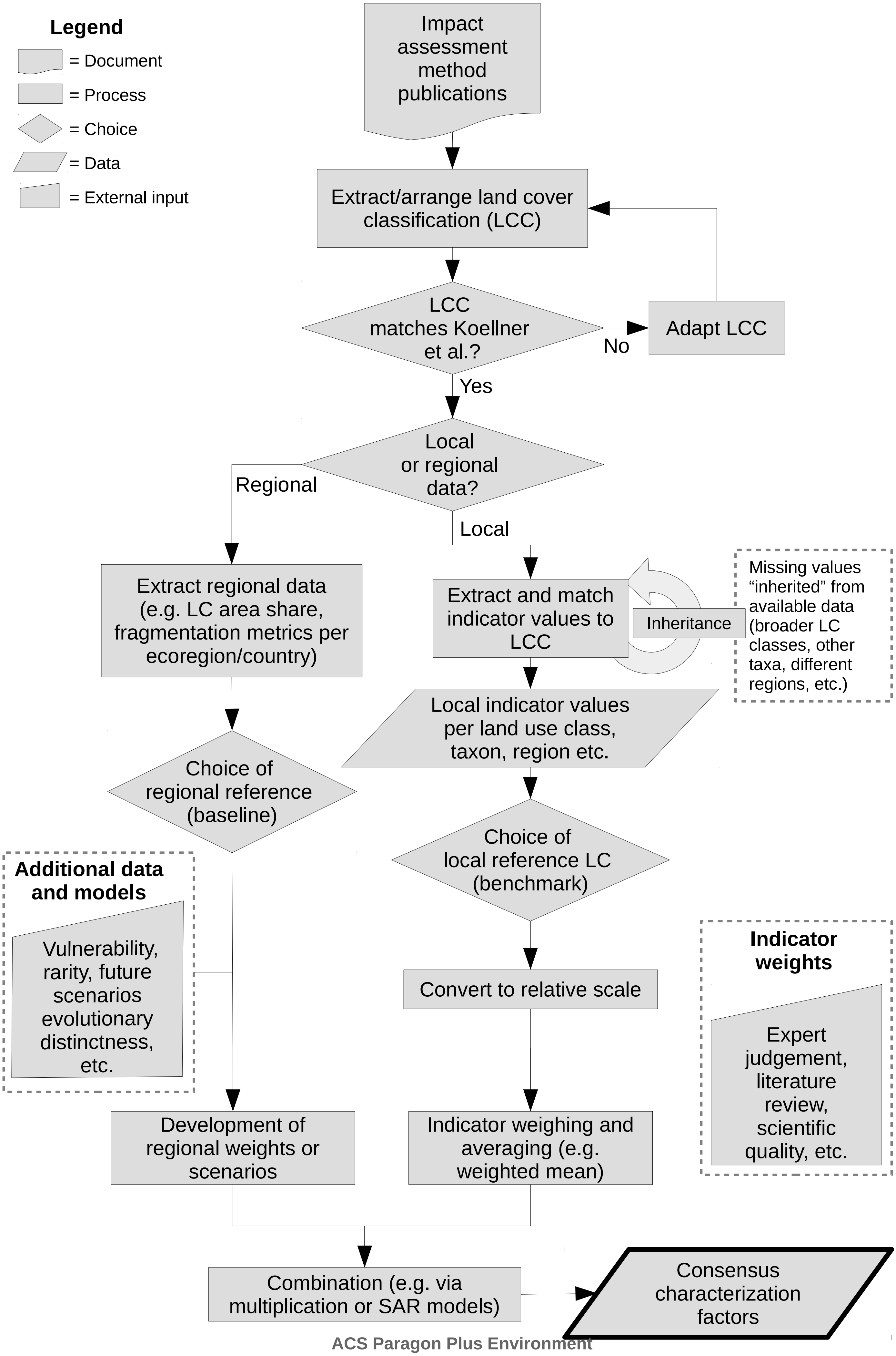
de Baan et al. (2013a)
Michelsen (2008)
Leh et al. (2013)
Coelho and Michelsen (2014)
Brentrup et al. (2002)
Lenzen et al. (2009)
Souza et al. (2013)
Koellner and Scholz (2007, 2008)
de Schryver et al. (2010)
Schmitt (2008)
Elshout et al. (2014)
Geyer et al. (2010a, 2010b)
Mueller et al. (2014)
Burke et al. (2008)
Urban et al. (2012)
Lindeijer (2000)
Category geometric mean


Legend
= Document
= Process
= Choice
= Data
= External input
Impact assessment method publications
Extract/arrange land cover classification (LCC)
LCC matches Koellner et al.?
No
Adapt LCC
Yes
Local or regional data?
Regional
Local
Extract regional data (e.g. LC area share, fragmentation metrics per ecoregion/country)
Extract and match indicator values to LCC
Inheritance
Missing values "inherited" from available data (broader LC classes, other taxa, different regions, etc.)
Local indicator values per land use class, taxon, region etc.
Choice of regional reference (baseline)
Choice of local reference LC (benchmark)
Additional data and models
Vulnerability, rarity, future scenarios evolutionary distinctness, etc.
Convert to relative scale
Indicator weights
Expert judgement, literature review, scientific quality, etc.
Development of regional weights or scenarios
Indicator weighing and averaging (e.g. weighted mean)
Combination (e.g. via multiplication or SAR models)
Consensus characterization factors