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Biophysical Measures to Support Analysis and Communication of Existence Values

James Boyd
Resources for the Future

Robert Johnston
Clark University, rjohnston@clarku.edu

Paul Ringold
Center for Public Health and Environmental Assessment

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James Boyd,[2] Robert J. Johnston,[3] and Paul Ringold[4]

[2] Resources for the Future, Washington, DC
[3] George Perkins Marsh Institute, Clark University, Worcester, MA
[4] US EPA, Office of Research and Development, Center for Public Health and Environmental Assessment, Pacific Ecological Systems Division, Corvallis, OR

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Abstract

A recent focus of ecosystem services research has been on the definition of biophysical outcomes and measures most closely linked to social welfare. There is a particular need to identify biophysical outcomes corresponding to existence values. (Values associated with existence apart from any current or future use). We review economic and ecological evidence to answer two key questions: First, what are ideal characteristics of linking indicators for existence values? Linking indicators should be: understandable, subject to direct sensory perception, represented at relevant temporal and spatial scales, comprehensive, and quantifiable in a repeatable manner. Second, what types of ecosystem outcomes are most likely to be associated with these values? We distinguish between indicators of taxa and ecological landscapes, and then multiple subcategories within each. Our fundamental conclusion is that while there are general principles informing the specification of linking indicators of existence values, there is no compact set of indicators or measures that applies universally. The case-specific nature of these issues—general guidelines notwithstanding—implies the need for sustained partnerships between social and biophysical scientists to address questions of indicator choice.

1. Introduction

A recent focus of ecosystem services research has been on the definition of biophysical measures that best represent the ways in which ecosystems contribute to social welfare or well-being (Boyd and Banzhaf 2007, Fisher et al. 2009, Johnston and Russell 2011, Boyd and Krupnick 2013, Boyd et al. 2016, Olander et al. 2018). These efforts are sometimes referred to as environmental "commodity definition." As discussed by Boyd et al. (2016), these definitions and measures support multiple purposes, including 1) effective communication of ecosystem service status and change to the public, 2) economic valuation of these changes, and 3) integrated assessments that model the production, consumption, and value of these services.

Because ecosystems are *systems*, virtually anything in nature can—at least in principle—have a direct or indirect effect on socially valued outcomes. Even something not valued directly can influence social welfare through causal impacts on other, directly valued goods and services.[5] Accordingly, almost everything in nature is *potentially* valuable—even from a purely anthropocentric and instrumental perspective. However, commodity definition emphasizes the identification of specific types of environmental features or conditions: namely, those *most closely or directly* linked to social welfare (Boyd et al. 2016, Olander et al. 2018). Various terms are used for these commodities in the literature, including "ecological endpoints," "final ecosystem goods and services," and "linking indicators" (the term used in this paper).[6]

This paper synthesizes economic and ecological evidence from the literature to propose

[5] An example is atmospheric GHG concentrations. They do not directly impact welfare (and are imperceptible except to scientific instrumentation). However, they affect temperature, precipitation, and risks of flood and fire, which do directly matter to welfare.

[6] See, for example (Boyd and Banzhaf 2007, Fisher et al. 2009, Johnston and Russell 2011, Boyd and Krupnick 2013, Boyd et al. 2016, Olander et al. 2018). These prior works also provide additional discussion of terminology.

principles and guidance for the development of linking indicators for one particular class of ecosystem service value, often called "existence value." In doing so, we attempt to answer the questions: (1) What are ideal characteristics of linking indicators for existence values?; (2) What types of ecosystem outcomes or services are most likely to be associated with these values?; and (3) What measures tracked by natural scientists are most likely to serve as valid linking indicators for these outcomes and services? Although there is widespread agreement that existence values (a subtype of nonuse values) can be present for many types of environmental outcomes, the empirical estimation of these values has been the subject of long debate and innovation within economics (Carson et al. 1999, Carson 2012, Kling et al. 2012, Johnston et al. 2017b). Much of this effort has been focused on the challenge of estimating the value itself, using stated preference (survey-based) methods, because these values are (by definition) unmeasurable using revealed preference or market data.[7] This paper mostly sets those valuation questions aside to focus on an equally important, under-appreciated, and corollary issue: *what are the biophysical features that give rise to—or are most closely linked to—existence value and how should we measure these features?* If we are to measure existence values, it is of obvious importance to ask: to *what* exactly do people attach such values and how can these things be most usefully measured?

Although past work has considered this topic briefly (e.g. Carson et al. 1999, Boyd et al.

[7] There is a vast economic literature going back to the 1970s on ways to monetize existence and other nonuse values. The majority of that literature centers around stated preference methods designed to simulate choices or behavior in hypothetical markets in order to derive values, or that elicit values directly. These methods have been improved to the point that state-of-the-art studies yield results broadly consistent with studies relying on real world behaviors (Kling et al. 2012, Johnston et al. 2017b), though some economists challenge this general conclusion and debate remains regarding the methodological validity of hypothetical preference as a proxy for preferences measured via actual behavior (Hausman 2012). Bishop and Boyle (2019) conclude that the "weight of evidence suggests that [stated preference methods have] sufficient reliability and validity to be a useful tool to inform policy analysis and litigation, although research should continue to improve the method."

2016), it has remained largely unexplored in rigorous terms. While there is an extensive literature that seeks to estimate nonuse and existence values for ecological outcomes, the biophysical measures used to quantify these outcomes vary widely, and there are no consensus guidelines on the desirable properties of these measures. Hence, analysts frequently struggle to determine "what to measure" when describing ecosystem goods and services most closely linked to prospective existence values. Even when the commodity type has been determined in general (e.g., the condition of a threatened or endangered species) it is not always clear how it should be measured with respect to the provision of existence values (e.g., official listing status, probability of continued survival, population size, species range, etc.), and what types of units should be used (e.g., cardinal versus relative units) (Ojea and Loureiro 2011, Johnston and Zawojska 2020).

Questions of this type have been addressed to some degree for other categories of ecosystem services, including those that give rise to "use values" such as those linked to human consumption, health, and recreation. These issues lie at the heart of international ecosystem service classification schemes designed to help government agencies, NGOs, the private sector, and academicians synthesize, interpret, and standardize ecosystem services analysis.[8] To date, however, there has been no concerted attempt to address the same commodity definition question for existence values.[9]

[8] The goal of this research area is to provide practical guidance on biophysical measurement to underpin a variety of analytical activities, including cost benefit analysis, ecosystem status and trends studies, and environmental accounting initiatives.

[9] Examples include the National Ecosystem Services Classification System (NESCS Plus) in the U.S. and the Common International Classification of Ecosystem Services (CICES) in Europe. As a broad category, existence value is represented in existing classification schemes. But very little guidance is provided on relevant commodities beyond a suggested focus on "emblematic" species (in the case of CICES). In its current form the NESCS Plus and its companion report on metrics (U.S. Environmental Protection Agency 2020) propose specific metrics, but those are based largely on expert opinion elicited in workshops. The reports themselves invite a more rigorous examination of metrics. Our analysis is intended to contribute to that more rigorous examination.

Common examples used to *illustrate* the concept of existence value include charismatic megafauna (e.g., marine mammals or charismatic fish such as Chinook salmon; Johnston et al. 2015), and iconic landscapes such as the Great Barrier Reef (Rolfe and Windle 2012) and the Grand Canyon (Kopp 1992). But even these examples raise questions about the linking biophysical measures that best represent and quantify the existence (versus non-existence) of these commodities from the perspective of existence values. For megafauna, should we focus only on extinction events (the existence of the *last* member of a marine mammal taxon), the probability of extinction in the foreseeable future, or something related to current population numbers or species ranges? Considering an iconic ecological landscape, what defines the Great Barrier Reef from the perspective of existence value? Is it the percentage of live coral cover, or is it something beyond that (e.g., the diversity of fish populations living there)? Does the reef continue to exist if the live corals perish but skeletons remain?[10] Can relevant attributes be quantified using a single variable, or are multiple measurements required? Similar questions arise when considering potential existence value associated with less iconic natural resources such as lesser-known species, or landscapes that are not nationally or globally iconic but perhaps appreciated at a local scale. A related challenge is that ecosystems such as these often lack clear geographic and biophysical boundaries (even to ecologists), complicating the spatial definition of existence measures.

Many types of policy analysis and guidance, including economic valuation, rely on an accurate quantification and communication of outcomes that *are valued directly* by different groups of people. Providing research and policy advice in terms of measurements that are only *indirectly* related to valued commodities may lead to invalid results. For example, mis-specifying

[10] From an ecological perspective this question might seem ridiculous. But some people may consider a reef structure valuable even if biologically dead.

linking indicators within stated preference research can lead to biased estimates of economic value (Johnston et al. 2017c). This observation is not new. In the early 1990s, Smith (1993) argued that "research must explore how to describe the public good services underlying nonuse values…" Limitations in ecologists' ability to influence public policy decisions has also been attributed to a communication problem in which "the language and format in which ecological information is presented to the public [is related to] aspects and processes of nature that have little interest or application to the public's concerns […]" (Norton 1998). Nearly three decades later, this remains an area in which guidance is lacking.

To address these questions, we begin with a brief introduction to the concept of existence value. This is followed by a discussion of relationships between existence values and linking indicators, focusing on the foundational question: The existence of *what*? We then explore the interdisciplinary literature related to three broad categories of ecosystem outcomes and services often associated with existence values—species, ecosystems, and landscapes.[11] Using these illustrative examples, we attempt to draw general conclusions regarding the types of existence-value indicators likely to be most useful for broader classes of ecosystem services.

2. Existence Values—A Brief Review

Within economics, existence values are not controversial as a matter of theory. The concept of existence values is routinely treated in academic textbooks and included in guidance for practitioners of economic analysis (Kopp and Smith 1993, National Research Council 2005, Freeman et al. 2014, U.S. Environmental Protection Agency 2014b, Champ et al. 2017).

[11] The following sections discuss indicators of ecosystems and landscapes within a single category denoted "ecological landscapes." The rationale for this categorization is described below.

Existence value may be thought of as one component of a broader category of values referred to by economists as nonuse values (or passive use values). From a theoretical perspective, nonuse values are often defined as "those portions of total value … that are unobtainable using indirect measurement techniques which rely on observed market behavior" (Carson et al. 1999). Nonuse values may also be described in more general terms as values that can be realized without any observable use or behavior related to the good or service (Kopp 1992), with reference to the seminal concepts of Krutilla (1967).[12] Davidson (2013) defines "non-use value as the benefit arising from knowledge that (part of) nature exists and will continue to exist, independently of any actual or prospective use by the individual."[13]

Given the emphasis of this paper on commodity definitions for existence values that are applicable across disciplines, we emphasize a conceptual definition. Specifically, this paper focuses on commodity definition as related to (pure) existence value, as defined by Davidson (2013): "the satisfaction of knowing that nature exists but not originating in altruism," where altruism includes bequest motivations. *Existence value (as defined here) is the value of simply knowing something exists apart from any other use, enjoyment, or benefit it may provide to ourselves or others.* This means that existence value is not the value of a resource that is consumed, like water, timber, or fish. Nor is it the aesthetic or recreational value of a landscape, nor the value of health improvements from cleaner air or water. Moreover, existence value is not the possible future option or bequest value of these "uses" to us or our descendants.

Existence values are easily integrated into and consistent with neoclassical welfare theory

[12] Economic theory does not provide a defensible means to empirically disentangle welfare measures as a function of underlying motivations—hence, describing nonuse value in terms of motivations (such as existence or bequest motivations) is not useful in terms of empirical value estimation. Nonetheless, motivation-oriented definitions continue to be used in the ecosystem services literature and can be useful for helping to define indicators or commodity definitions that may be most closely linked to these values (Smith 1993).

[13] Existence value is not the same as intrinsic value, to the extent that the latter is defined broadly as a value held by nature itself, apart from human welfare (c.f. Turner 1999, Davidson 2013).

and utility-based valuation frameworks (Smith 1987, Smith 1993). Structurally, nonuse values (including those related to pure existence motivations) may be defined as the difference between total value and use value (Smith 1987, Carson et al. 1999). Although some argue that existence values should not qualify as true economic values, "the standard view in economics is that decisions about what people value should be left up to them," and that existence values thus have the same welfare-theoretic basis as any other type of economic value (Hanemann 1994). As a subset of welfare-theoretic nonuse values, the only requirement for a pure existence value to be "real" is that—independent of the other kinds of value it might generate—a resource's existence in a particular state or condition is preferred to its non-existence, or existence in another state or condition. This characterization clarifies that existence values may be held for outcomes that extend beyond the simple dichotomy of existence versus non-existence to the *state or condition* in which something exists.

Given a broad consensus that existence values are real and potentially relevant, the primary focus of economic inquiry and debate has been on the magnitude of existence values and the validity of methods used to measure them. This literature is entwined with the broader literature on stated preference methods, as these methods are the only generally applicable valuation approach able to measure existence values. This work not only finds evidence of existence values, but also that they can be of comparable (or larger) magnitude than other categories of environmental value, particularly in the aggregate (e.g., Stevens et al. 1991, Hanley et al. 2003, Johnston et al. 2003, Johnston et al. 2005a). Also worth noting is conventional opinion polling that suggests widespread "existence preferences" for certain natural resources (Kotchen and Reiling 1998, 2000, U.S. Department of Agriculture Forest Service and University of Tennesee 2000, Whitehead and Chambers 2003, Wainger et al. 2018).[14]

Our attention to the commodities that generate existence values is also motivated by the relevance of existence value estimation to public policy.[15] The validity of existence values as a component of natural resource damage assessments has been affirmed under U.S. federal law (US Court of Appeals 1989) and agencies' codified methodologies for damage assessment feature explicit reference to these values (Department of the Interior 2008, National Oceanic and Atmospheric Administration 2014). Existence values are often addressed (though usually not quantified) by Federal Regulatory Impact Analyses.[16] Measurement of existence values is also recognized as a legitimate component of Federal Cost Benefit Analysis methodology.[17] Beyond the U.S., existence values are a recognized component of environmental planning and evaluation frameworks in many other countries.[18] Finally, species protection laws, such as the Endangered Species Act (ESA), and international treaties, such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), provide political evidence of a social desire to protect species' existence. Notably, these laws and treaties are not focused solely on species with commercial, charismatic, and recreational value – they also protect more obscure, relatively unknown species. Hence, regardless of whether one accepts the validity of economic valuation, there is broad evidence that existence values are accepted and policy relevant.

[14] For example, surveying Maine residents, Kotchen and Reiling (2000) found that 70-80 percent of respondents felt it "important" or "very important" to protect Peregrine Falcons and Shortnose Sturgeon even if no one were ever to see the species.

[15] The use of existence value measures in public policy is not uncontested. Some economists question the accuracy and validity of existence value estimates on empirical and methodological grounds (Donald and Robert 1992). Philosophical critiques argue that existence values reflect ethical motivations outside economics' reach. According to one: "The theological problem with existence value is that it attempts to answer a religious question by applying an economic method" (Robert 1997).

[16] For example, see US Environmental Protection Agency (2014a)

[17] See Office of Management and Budget (2003) and US Environmental Protection Agency (2014b)

[18] As one example, see ten Brink et al. (2013).

3. Properties of Existence Value Linking Indicators

It is common for environmental features or conditions to give rise to both existence values and other types of value, including use values. Accordingly, this paper does not focus on linking indicators connected *solely* to existence values. However, when describing linking indicators, we emphasize the properties required for a measure to be suitable for assessment of existence values. Although similar properties *might* enable the measure to serve as a linking indicator for other types of value, we do not consider that issue here.

The fundamental characteristic of linking indicators, defined with respect to both ecological and economic production, is *direct welfare relevance to at least one human beneficiary* (i.e., they must matter to people directly). As such, linking indicators represent the preferred biophysical measures to underpin ecosystem services communication, classification, accounting, and valuation (Boyd et al. 2016). They also represent ideal points of contact between natural and social science methods and data. Given that linking indicators serve as an interface between social and biophysical analysis, it follows that their validity must extend into both realms. Hence, in the discussion that follows we evaluate candidate commodities along two dimensions. First, we seek indicators that are valid from a social science perspective, in that they facilitate accurate quantification and understanding of ecosystem services for purposes such as classification, accounting, valuation, and communication. Second, we seek indicators that are valid from a natural science perspective, and that can be measured consistently across geographic scales.

A. Formal Definition: Linking Indicators

As noted earlier, the focus of commodity definition is on a specific type of environmental feature or condition: namely, those most closely linked to social welfare. We refer to these features or conditions as linking outcomes. Linking outcomes delineate, conceptually, an aspect of the natural world that provides direct human benefit (i.e., also called a final ecosystem good or service). For example, "fish abundance" might be a linking outcome for species that are valued by people. However, linking outcomes are not direct empirical measures. Hence, a subsequent step is required to render a linking outcome measurable and hence useful for empirical analysis.

This subsequent step is the linking indicator. Linking indicators are empirical metrics or measures of linking outcomes. Multiple metrics (or linking indicators) may be available for each linking outcome. For example, if fish abundance is the linking outcome, the empirical measure or metric of fish abundance used for analysis is the associated linking indicator. As an illustration, for the abundance of a migratory species such as Atlantic salmon (*Salmo salar*), one possible measure (the linking indicator) of fish abundance (the linking outcome) might be the number of adults returning upstream to spawn in a given river, over a particular period of time. Because of the close relationship between linking outcomes and linking indicators (one simply being an empirical measure of the other), the terms are often used interchangeably.

Linking outcomes are components of an *ecological production framework*. These frameworks—sometimes called ecosystem service causal chains (Bell et al. 2017, Olander et al. 2018)—depict causal linkages among biophysical outcomes in an ecological system and can be thought of as networks of ecological features linked by biotic, hydrologic, chemical, and other biophysical processes. They are common in ecological theory and practice (e.g., food webs, water cycle models, or spatially explicit population models). For example, a model that relates biotic and physical conditions to species' abundance – via competition, predation, migration, and reproduction processes and how those factors vary with habitat– depicts a production framework. They are also analogous to input-output frameworks in economic accounting and productivity analysis (Dietzenbacher and Lahr 2004). Production frameworks allow us to distinguish between inputs and outputs via chains of causation, with indicators used as the associated empirical measures.

Linking outcomes are a subset of these outputs. Specifically, linking outcomes are biophysical features that *directly* affect people's welfare. The associated linking indicators may be field measurements, or modeled quantities or qualities; ecological metrics or indices are specific types of linking indicators. While many, if not all, biophysical outcomes play a role in our wellbeing, not all biophysical outcomes *directly* influence social welfare.[19] For example, nitrogen concentrations in water are important to many, but typically only indirectly (i.e., as an indirect input to linking outcomes). Usually, we do not care about this feature as an end itself (Johnston and Russell 2011). Rather, we care about it because of the outcomes it affects (e.g., fish abundance, hazardous algal blooms, or water clarity). If the goal is to communicate and socially evaluate the implications of ecological change, more direct outcome measures are preferred. Few individuals have the expertise or data to accurately translate indirect biophysical outcomes to the biophysical outcomes that drive social welfare (Johnston et al. 2017c). Hence, even when people are aware of a causal connection between a direct and indirect outcome (e.g., nitrogen and fish), they are unlikely to understand the relationship accurately or quantitatively. Thus, linking outcomes are important to public communication and quantitative social evaluation of ecological change.

[19] The definition of what "directly matters" is not always clear cut and is complicated by the myriad ways people enjoy, use, and benefit from nature. For a broader, more detailed treatment of the issue, including the analogy to final and intermediate goods in economic accounting, see Boyd et al (2016) and Johnston and Russell (2011).

From an economic standpoint, the value of indirectly valuable inputs is embodied in the value of a resulting outcome that directly matters to welfare, i.e., the linking outcome.[20] Moreover, that value can be quantified once the value of the linking outcome is established – if the ecological production function between the input and output is known (e.g., the relationship between nitrogen, dissolved oxygen, and fish abundance). Also, indirect outcomes can be important to the management of natural resource systems, where they are used as "leading indicators" of environmental change. Accordingly, a focus on linking outcomes does not imply that the value of other ecological inputs is overlooked or irrelevant. To the contrary, establishing the value of linking outcomes emphasizes and helps quantify the value of indirectly valuable ecological inputs. Moreover, if we seek to understand or manage the distribution, abundance or qualities of the linking outcome then understanding, measuring, and managing intermediate or indirect factors is essential.

Grounded in this underlying definition, linking outcomes for existence values are simply linking outcomes that generate one type of value—existence value as defined above. The same outcomes may simultaneously generate use values. However, the defining characteristic of linking outcomes for existence values is that value is still provided, even if no other values (e.g., use values) are present.

B. Summary and Additional Indicator Criteria

Based on the above definition, the core criterion for a linking indicator is that:

1. The indicator should measure a biophysical outcome that affects social welfare as directly as possible.

[20] Our terms "inputs" and "linking outcomes" correspond to "intermediate" and "final" goods and services in economic accounting, where final goods are what is consumed by end users and where the value of final goods embodies (includes) the value of inputs used to produce them.

a. Social welfare includes the welfare of specific beneficiary groups along with that of society as a whole.[21]
b. Outcomes that directly matter are those valued for their own sake, as opposed to outcomes valued as inputs to, or proxies for, other outcomes that directly matter.
c. Ideally, the indicator should be identified as a linking indicator based on some type of empirical evidence (e.g., rather than solely based on thought experiments).[22]

We also posit several corollary criteria drawn from the literature on factors biophysical scientists use to specify ecologically meaningful and useful biophysical indicators (Jackson et al. 2000, Dale and Beyeler 2001, Niemi and McDonald 2004, van Voorm et al. 2016)[23] and a literature on criteria for biophysical features to be used in social analyses (e.g. Johnston et al. 2012, Schultz et al. 2012, Zhao et al. 2013, Boyd et al. 2016, van Voorm et al. 2016).

2. The indicator should be presented in a way that is understandable to lay audiences. Often, this means that use of technical terms and jargon are undesirable.[24]
3. All else equal, measures of outcomes that can be sensed or experienced directly (i.e., seen, smelled, heard, tasted, and touched) are more likely to serve as valid linking indicators than those linked to outcomes that are entirely beyond our sense experience (a category that includes many outcomes derived from scientific instrumentation).[25]
4. The indicator should be measured, or modeled at a temporal and geospatial scale linked to the existence value in question.

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[21] Interactions with nature, and the ways nature is perceived as valuable, vary widely across social groups. Accordingly, there is not necessarily a single, uniform set of linking indicators of equal direct relevance to every social groups. We stress only that a linking indicator be directly relevant to at least one such group of people.

[22] We elaborate on such empirical evidence later in this section. Several types of empirical evidence can provide insight, including results from the social science literature (e.g., stated preference or focus group results) or evidence from review of statutes or regulations that emphasize certain biophysical outcomes.

[23] Four general criteria emerge from this literature: (1) interpretability or salience, (2) comprehensiveness, (3) usefulness in quantitative analysis, (4) and feasibility of measurement or estimation.

[24] This does *not* imply that only simple or single-metric indicators can serve as linking indicators. Rather, it implies that linking indicators—regardless of their underlying complexity—must be comprehensible in common-language terms.

[25] This last guideline may seem to conflict with our interest in existence values. After all, existence values relate to things we will never see or touch. However, just because we won't see or touch them does not mean we couldn't see or touch them; and in our mind's eye understand them with the same tangibility.

These criteria relate to the interpretability or salience of the indicator. Interpretability is a central criterion from our perspective. An indicator requiring expert or technical explanation to make its meaning clear is less useful than one whose meaning is more directly understood, particularly when indicators are meant to be useful for both biophysical and social/policy analysis. An emphasis on indicators that are understandable (salient and interpretable) to lay audiences emerges from the literature devoted specifically to the characteristics and validity of linking indicators[26] within ecosystem services assessment (e.g. Johnston and Russell 2011, Boyd and Krupnick 2013, Boyd et al. 2016, Olander et al. 2018).

These criteria are also supported by the literature devoted to communication of ecological information to the public (e.g. Norton 1998, Schiller et al. 2001). For example, this literature argues that indicators expressed solely in technical or scientific terms are unlikely to be ideal linking indicators, because the relevance of these indicators is not broadly understood. To address this problem, this literature calls for effective "bridge concepts to create indices that are, and should be, of interest to concerned citizens" (Norton 1998). Linking indicators are designed to serve as these bridge concepts.

Another important aspect of interpretability is that an indicator reflect the temporal and spatial dimensions of the outcome being measured in ways that are themselves clear and understandable to lay audiences. For example, it may be important not just to measure a species' abundance, but to communicate where specifically that abundance occurs on the landscape and over what period.

5. The indicator should contribute to a comprehensive understanding of ecological

[26] Or similar concepts with alternative names, such as final ecosystem service indicators or benefit relevant indicators.

conditions.

Comprehensiveness, an important indicator criterion in the ecological science literature, means that an indicator (or collection of indicators) captures a breadth of relevant features or qualities. For example, if the goal is to capture "air quality," a representation of ozone concentrations alone is far less comprehensive and relevant than a representation of multiple air quality constituents. Comprehensiveness will be important to our search for linking existence value indicators because people often value the existence of compound environmental conditions, as opposed to more narrowly defined features or qualities.[27]

6. The indicator should be quantitative and repeatable.

Linking indicators are used to facilitate subsequent social analysis. Hence, an indicator should be defined with sufficient clarity that the certainty and interpretation of the quantification across different studies, geographies, or time periods can be communicated to users. This information can help users determine if the measure's certainty is sufficient for their needs. Some types of indicators are more likely to serve this purpose than others. For example, purely narrative indicators are less desirable linking indicators. Continuous indicators are usually preferred to categorical variables derived from an underlying continuous distribution, except in cases where the derived categorization is (a) based on clearly defined and objective features of the data, and (b) conveys information that is obscured by the underlying continuous distribution and is directly relevant to existence value.

In many cases, linking indicators are derived from an underlying continuous

[27] There can be a tradeoff between comprehensiveness and interpretability, as indicators that capture a greater breadth of ecological conditions may be more difficult for laypersons to interpret. In such cases, an optimal linking indicator strikes the best balance and in a way that most closely matches what people most directly value and understand.

measurement or model estimate. Recognizing this fact, continuous measurements (or model estimates)—assuming they meet the first five criteria for linking indicators described above—can often serve as superior linking indicators, because they provide richer and more complete information than the categorical or discrete variables that might be derived from the original continuous measurement. This is particularly true when derived categories have no clear objective interpretation when abstracted from the underlying continuous variable.

For example, consider an indicator that categorizes an ecosystem state as good, fair, or poor (a categorical indicator). Absent an underlying continuous representation, an ecosystem change may not trigger a conversion of the indicator from good to fair, yet that change may nevertheless be meaningful in economic or other terms. In other words, a continuous representation (even if it is used to construct a categorical measure) captures change more usefully because it enables more sensitive analysis. Another advantage of continuous data is that categorization typically requires a transformation (or translation) of the continuous scale (e.g., 0-100) to derived categories (e.g., low = <25, medium = 25-50, etc.). If that translation varies for different beneficiaries or demographic groups or in different times and places, then the existence of continuous data allows for the application of different context dependent translations. The challenges of translation are magnified if the translation involves a normative judgement not implied by the underlying continuous variable (for example whether something is "good").

There are exceptions to this general rule. In some cases, the categorization of a continuous measurement provides information that is otherwise obscured and is relevant to existence value. An example is the designation of a species as "endangered" or "critically endangered" according to a set of objective and non-normative criteria, based on an underlying population estimate. This *does* require a further translation of underlying continuous measurements, as explained above. However, in cases such as this, the translation and categorization can provide additional information that may be relevant to understanding by non-experts, above and beyond information available from an underlying continuous variable. For example, informing a non-expert that North Atlantic Right Whales are endangered might provide more useful and interpretable information than an underlying estimate of 336 whales in the population.[28] Hence, assuming that a categorical variable is well defined, the primary consideration for whether a categorical or continuous variable is a preferred linking indicator is the extent to which each provides *directly interpretable and relevant information* to non-experts—and particularly information relevant to their existence value.

However, as discussed by Olander et al. (2018) for more general applications, it is rarely the case that purely narrative, qualitative descriptions will serve as suitable linking indicators, absent the underlying continuous information. Although narrative descriptions can be informational when presented in concert with well-defined linking indicators, they lack the resolution, repeatability, and quantitative nature to serve as linking indicators themselves.

We conclude by noting that the identification of linking indicators is an iterative process in which social scientists help to broadly identify the kinds of environmental features whose existence is important to individuals or beneficiary groups (i.e., linking outcomes). Biophysical scientists can then respond with options for how to estimate and represent those biophysical features (i.e., the associated linking indicators). Then biophysical and social scientists can work together to determine which of the options best meets the needs of a particular analysis.

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[28] This argument is similar to that provided by Johnston et al. (2012) for the presentation of indicators in *both* cardinal and relative terms. As explained by Johnston and Zawojska (2020, p. 1246), "differences that might seem large in cardinal numbers (e.g., 2,000 versus 20,000 birds) might appear trivial when viewed in relative terms (e.g., "much less than 1% of the population" versus "less than 1% of the population"; Desvousges et al. 1993; Hanemann 1994; Carson 2012, p. 34)."

C. Evidence to Support Commodity Definition

In what follows we call upon the economics literature – primarily the environmental stated preference literature – to help identify and illustrate good candidates for existence value linking indicators. Before doing so, it is important to review both the strengths and limitations of that literature as a source of evidence for the definition of existence commodities.

The extensive stated preference valuation literature derives environmental value estimates based on survey responses (Johnston et al. 2017b, Hanley and Czajkowski 2019). These methods ask subjects to consider (and often choose among) hypothetical but feasible environmental scenarios based on their preferences, in a way that allows the researcher to quantify preferences (often in monetary terms) and often evaluate the extent to which subjects are willing to make tradeoffs between different types of outcomes. In many cases, existence (or broader nonuse) values are a substantial component of the total willingness to pay (WTP) or willingness to accept (WTA) estimates generated by these efforts (Carson et al. 1999, Hanley et al. 2003, Johnston et al. 2003, Johnston et al. 2005b). Importantly, stated preference surveys often require researchers to develop and present environmental or ecological outcomes that are closely related to existence values so that these values are captured by survey scenarios and the value estimates derived from these scenarios.

Best practice in this literature calls for the use of focus groups or interviews with respondents drawn from the target population to develop scenarios that are understandable and meaningful to the study's subjects (Johnston et al. 2017b). Among the goals of these focus groups and interviews is to obtain insight on how to define and communicate environmental commodities effectively (Johnston et al. 1995, Kaplowitz et al. 2004, Weber and Ringold 2015, Weber and Ringold 2019). Therefore, assuming that best practices for survey development are applied, indicators used in stated preference surveys *should* ideally reflect deliberate input from non-expert subjects on environmental commodities they perceive to be understandable, relevant, and valuable. In addition, a few studies have conducted systematic empirical evaluations of the extent to which alternative types of indicators in these surveys lead to valid or robust expressions of value (e.g. Johnston et al. 2011, Zhao et al. 2013) and have proposed specific guidelines for indicators used within these studies (Johnston et al. 2012, Schultz et al. 2012).

As a result, review of stated preference studies can provide at least some insight into the types of indicators associated with existence values. That said, this form of evidence should be treated with caution. One reason is that the focus of such studies is almost always on the valuation exercise, rather than commodity definition per se. Another reason is that the choice of commodities and indicators in such studies is ultimately under the control of the researcher and can reflect methodological considerations beyond a search for well-defined commodities. For example, researchers may not have the data or expertise to quantify "ideal" ecological indicators and may therefore rely on less ideal but more available alternatives (Johnston et al. 2012). In other cases, practicalities or predetermined research goals may constrain researchers to use certain commodities or indicators, even if focus groups do not suggest that these are ideal choices for survey design. As a result, despite attention to survey design and testing, many published studies in the stated preference literature have used indicators that violate key conditions for validity (Johnston et al. 2012, Schultz et al. 2012) and that do not adhere to the criteria we identify above. Despite these qualifications, stated preference studies represent an important window onto existence value commodity definition because lay audiences (study subjects) are explicitly asked to reflect on the environmental conditions and features they understand and view as important.

4. The Existence of *What?* Taxa (Species) Versus Ecosystems and Landscapes

For purposes of discussion, we propose two broad, illustrative categories of commodity for existence values linked to living natural systems: (1) indicators for taxa and (2) indicators for ecosystems and landscapes. These are not the *only* possible ecosystem-related sources of existence value. Rather, these illustrative examples are used to characterize measures and metrics likely to serve as linking indicators in broader contexts. We also propose these categories because they include some of the most common types of commodities linked to nonuse and existence values in the ecosystem services literature. These categories capture existence value of species (taxonomic outcomes) and broader systemic outcomes (e.g., the existence of an ecosystem or natural landscape), among other valued ecological outcomes. The two categories are defined in section 4A and 4B below. Figure 1 summarizes our categorization of indicators and provides a guide to our analysis below. [INSERT FIGURE 1 HERE]

A. Taxa

This set of linking indicators captures the existence and condition of living organisms or species, considered either individually (e.g., an individual platypus species) or in clearly delineated groups (e.g., all species within the family *Ornithorhynchidae* or an assemblage of taxa within an ecosystem). The use of taxa-related linking indictors poses fewer definitional issues than indicators for landscapes and ecosystems because we can rely both on clear, existing taxonomic classifications and on lay audience comprehension of related concepts. For example, the definition of a "species" (i.e., a collection of individuals capable of interbreeding) is relatively unambiguous and corresponds to public interpretation of the term.[29] If people hold an existence value for species X (e.g., the pygmy rabbit, *Brachylagus idahoensis*), it is relatively easy to define and bound the meaning of this concept. The broader classification term "taxa" reflects the fact that the relevant grouping for certain types of value may not be solely in terms of individual species but also other levels of taxonomic classification such as a family of species. Among the key questions for linking indicator definitions in this category are the characteristics of each organism or group that are (most) closely linked to existence values and the spatiotemporal scales over which indicators should be measured. Other questions relate to the most relevant grouping of organism types. For example, in some cases regional subspecies might be relevant for existence values (e.g., Northern versus Southern White Rhino), whereas in other cases the most relevant taxa might be a larger grouping.

B. Ecosystems and landscapes:

While evidence suggests that people hold existence values for ecosystems and landscapes, this category poses more significant definitional challenges for linking indicator development. The concepts of "an ecosystem" and "a landscape" are different, and both can be difficult to define in precise terms that are accessible to non-experts (e.g., how does one define a "landscape" for purposes of existence value?). However, in terms of linking indicator development, they share common properties. First, both represent collections of heterogeneous features. In other words, they are *composite* commodities requiring indicators that reflect their composite nature. Second, both involve geographic delineation – an "area," a "large area," a

[29] In using "relatively" we note the vital discussions on the definition of species in the biological literature (e.g. Frankham et al. 2012, Wilkins 2018).

"region" – associated with the ecosystem or landscape definition. Third, both concepts suffer from a lack of clear operational consensus around precise geographic boundaries. For these reasons we treat indicators of ecosystems and landscapes as a single category of indicator – a category we will refer to going forward as *ecological landscapes*.[30] (The issue of ecosystems' and landscapes' geographic boundaries is discussed in more detail below.) Also, we hypothesize that the lay public is likely to equate the two concepts (landscape and ecosystem). In other words, an existence value expressed for a landscape is likely to also be expressing its value as an ecosystem.[31]

5. Taxa—Evidence and Proposed Indicators for Existence Values

A taxon (plural taxa) is defined as a collection of organisms of any taxonomic rank (e.g., species within a genus or family).[32] There have been many stated preference studies that estimate WTP (including estimates of both use and nonuse values) for changes in species' abundance, distribution, or extinction risk. In many cases, direct uses of these flora or fauna are minimal, such that a substantial proportion of WTP is comprised of existence or other types of nonuse value. Some, though not all, of this literature focuses on the nonuse value of threatened and endangered species or taxa.[33] This work provides evidence on the type of measures that can

[30] In fact, some ecological definitions of landscape are based on ecosystems. For example, Forman and Godron (1981) define a landscape as "a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout."

[31] Although we recognize that there are exceptions to these generalizations, this grouping allows us to proceed with discussions of indicator properties without having to first struggle with the likely intractable challenge of clearly disentangling differences between ecosystems and landscapes.

[32] We use this term rather than something more restrictive such as "species," because individual species are not always the desired "units of management" or public perception. For example, the units of management of Pacific Salmon in the northwestern US relate to hundreds of individual stocks (Nehlsen et al. 1991, Pacific Fishery Management Council 2020).

[33] For example, stated preference surveys have attempted to quantify the value of changes to threatened, rare, or endangered species or taxa such as salmon, silvery minnow, whooping crane, bald eagle, striped shiner, gray wolf, squawfish, arctic grayling, Mexican spotted owl, Northern spotted owl, Steller sea lion, monk seal, bottlenose dolphin, northern elephant seal, gray, humpback, and blue whales, sea otters, bighorn sheep, peregrine falcon,

serve as effective linking indicators for taxa. As discussed by Johnston et al. (2012) and Schultz et al. (2012), however, the inclusion of an indicator in past stated preference studies does not ensure the quality of that indicator—many past studies have included indicators that lack key requirements for social and ecological validity. Nonetheless, patterns from the literature can provide insight into the types of indicators considered to be useful in past work.

In principle, it is possible for people to hold existence values for almost any type of change in a taxon (e.g., increase in a species' abundance or distribution). However, in practice, most efforts to elicit nonuse or existence values for taxa may be organized around two broad categories of value: (1) the value of avoided extinction and (2) the value of abundance independent of extinction risk. We therefore orient our discussion around these categories, defined broadly. For both categories of value, indicators can be either deterministic (e.g., population size or density, presence/absence) or probabilistic (e.g., extinction probability or a population viability measure). Also, the geographic range associated with binary existence or abundance measures need not be global. Spatial characteristics are relevant to both types of measures because existence values can depend on *where* this existence or abundance occurs (Glenk et al. 2020).[34] We further distinguish between values related to individual taxa and those related to (multi-) *species diversity and composition*, for example reflecting biodiversity or the prevalence of invasive versus native species (Jacobsen et al. 2008; Dissanayake and Ando 2014). Because the latter measures reflect composite, heterogeneous assemblages of taxa within ecological systems, we consider these under the ecosystems and landscapes category discussed later.

steelhead, red-cockaded woodpecker, fairy shrimp, wild turkey, and sea turtles, among others. Richardson and Loomis (2009) summarized this literature and provide bibliographic references.

[34] For example, Johnston et al.(2015) demonstrate distinct nonuse values for Chinook salmon in different US river systems.

The value of avoided extinctions clearly relates to existence value since extinction extinguishes that value. This category of value we refer to as the value of "binary (yes/no) existence." Associated indicators address questions such as "is the taxa extinct?" or "what is the probability of extinction?" However, the concept of existence values does not imply that value must relate solely to binary existence. There is abundant evidence that existence values may be held for changes in abundance that do not affect extinction risk.[35] For example, Bulte and Van Kooten (1999) consider this question for minke whales, demonstrating the importance of distinguishing between the non-use benefits of preventing extinction and the added marginal benefits of preserving numbers above the minimum viable population. Hanley et al. (2003) and Zhao et al. (2013) show that individuals are willing to pay for changes in the abundance of birds and fish, respectively, regardless of whether those changes are framed in terms of extinction risk or population changes with no mention of extinction risk.[36]

The following sections suggest specific linking indicators associated with these two categories of value, based on a review of the related literature. The goal is not to list a few indicators that apply universally, but rather to provide insight into the type of measures likely to serve as linking indicators for taxa-related existence values within these two general categories.

A. Indicators of Existence and Existence Probability

We first consider binary existence measures. These indicators may consider either global status (does a taxon exist anywhere?), existence over a specific spatial range (does it exist in a

[35] We emphasize that abundance indicators are relevant to binary existence values as well, since abundance measures related to extinction thresholds can signal extinction risk.

[36] Hanley et al. (2003) provide these WTP estimates for the general public, most of whom are nonusers of the birds in question, as well as for direct users. Zhao et al. (2013) address migratory fish that have no direct use in the studied area. Another explicit example is provided by (Johnston et al. 2005b).

given location?), or existence in a non-spatial context (does it exist in the wild?). These types of indicators are simple and have been shown to be directly relevant to nonuse value. A related indicator is the *probability* of these binary events.[37] Economic theory and intuition suggest that if binary occurrences matter to welfare, then their associated probabilities are also relevant. Indeed, one might argue that probabilistic treatments such as these are more useful linking indicators than binary treatments, because they capture a wider range of possibilities: not just the possibility that an extinction occurs with certainty. They are also more relevant from a policy and applied valuation standpoint because future conditions are rarely certain. In addition, the technical analyses providing estimates of current and future existence of populations or species provide probabilistic and relativistic estimates rather than binary ones (e.g. Morris et al. 1999, Beissinger and McCullough 2002, Morris and Doak 2002). Hence, the existence value estimates that are most relevant for real-world policy evaluation are those that consider effects of changes in extinction probability—not those associated with certain existence versus extinction.

Many indicators in past stated preference studies can be interpreted as versions of the binary extinction commodity: "Preservation of the species" (Bowker and Stoll 1988, Whitehead 1992); "protection of a stable population" (Whitehead and Chambers 2003); "achieving a self-sustaining breeding population," (Kotchen and Reiling 2000); "avoidance of local extinction" (Stevens et al. 1991);[38] "recovery of a species to its minimum population necessary to prevent extinction" (Ojea and Loureiro 2009), the number of "red-listed species on the heath [that] will be preserved" (Strange et al. 2007), and number of endangered species present (Lehtonen et al. 2003).[39] Finally, some studies combine regional existence indicators for multiple taxa into

[37] The probability of a realized extinction is equal to 1.

[38] *Local* extinction is obviously not equivalent to true, global extinction.

[39] The Ojea and Laureiro study also distinguishes between that commodity and "increases in population above the minimum necessary level" and confirm via valuation results the hypothesized difference between the

composite indicators reflecting the number of taxa present, such as the number of "endangered and protected species present" in wetlands (Morrison et al. 2002) or number of native species (e.g., native bird or fish species) that exist in an area (e.g. Morrison and Bennett 2004). Such indicators can be interpreted as measures of binary existence presented across multiple taxa (e.g., X individual species exist).

Although indicators such as these are at least superficially straightforward and linked to binary extinction probability, one must also consider whether they are well-defined from a biophysical perspective. For example, does "protection of a stable population" have an unambiguous, quantitative biophysical interpretation? Is the "minimum population necessary to prevent extinction" meaningful in biophysical terms? In some cases, descriptions such as these that might seem meaningful to non-experts can lack the properties of a well-defined biophysical indicator. In general, indicators (often verbal descriptions) that imply binary existence changes can only be considered valid linking indicators if they are grounded in well-defined biophysical metrics or measures with unambiguous interpretations.

Many stated preference studies also present probabilistic indicators related to species existence or extinction probability: "Improved percent chance of survival" (Rubin et al. 1991, Reaves et al. 1994); Change in "probability of future availability" (Brookshire et al. 1983); "Decrease in likelihood of extinction" (Stanley 2005). Some do not explicitly reference probability but implicitly embed it via reference to uncertainty: "increase in population with no guarantee of population recovery" (Giraud et al. 2002). Other studies use related population viability indicators—defined roughly as the probability that a taxon will continue to exist in an area as of a given time in the future (Johnston et al. 2012, Zhao et al. 2013, Johnston et al.

value of protecting against extinction and the value of population increases above that. See also Bandara and Tisdell (2005).

2017b).

An important feature of the probability-related commodity measures described above is their explicit or implicit reference to extinction-relevant thresholds. Extinction and population viability thresholds are biophysical concepts, associated with conservation biology, that relate population levels and habitat availability to a species' ability to reproduce at a rate sufficient to overcome natural or human-driven mortality (Groom et al. 2006, Gerber and González-Suárez 2010). Above these thresholds, changes in species abundance are in principle relevant to extinction risk (all else equal, the higher a species' population number the lower its extinction risk). But population measures anchored around these thresholds are particularly relevant because they identify step changes in extinction probability. Thus, if identifiable, classification of species in reference to whether they are below, at, or above such a threshold can serve as a useful extinction probability measure.[40]

Here again, biophysical indicator properties are relevant. For example, without additional underlying biophysical detail, an indicator such as "increase in population with no guarantee of population recovery" is not well-defined from a biophysical perspective. As emphasized by Johnston et al. (2012) and Schultz et al. (2012), many past stated preference studies have sacrificed unambiguous biophysical definitions to develop simple verbal indicators that are—at least superficially—easily understood by laypersons. However, unless these indicators are linked to precise and quantifiable biophysical outcomes, they lack the properties to be considered valid

[40] One challenge is that the categorical definitions used by different categorization systems are not equivalent. For example, the IUCN distinguishes between critically endangered, endangered, vulnerable, and near threatened, whereas the FWS distinguishes only between endangered and threatened. In principle, the IUCN's more granular categorizations are preferable as they allow more incremental extinction probability changes to be reflected. The IUCN also addresses a larger number of species. For a comparison of species coverage under the two systems see Harris et al. (2012). This in no way should be construed as a weakness of this literature. In practice, however, U.S. agencies and practitioners may prefer reliance on the more familiar and perhaps policy relevant FWS classifications.

linking indicators.

Thresholds such as these are implied in studies that communicate extinction probabilities for valuation purposes implicitly via official listing status, such as designations of endangered, threatened, etc., (Strange et al. 2007, Lew et al. 2010, Lew and Wallmo 2011, Johnston et al. 2012). Although indicators of this type are not strictly quantitative and can be influenced by official decisions on listing status, they are (arguably) easier to grasp by laypeople. Moreover, there is evidence that the lay public may have difficulty understanding probabilities expressed in traditional percentage formats, implying that such indicators—while quantitative—may not always be the best option for public communication and valuation (e.g. Slovic 1987, Lipkus et al. 2001, Edwards et al. 2002, Patt and Schrag 2003, Gilboa et al. 2008, Baker et al. 2009). Hence, while existence probabilities are undoubtedly relevant, there can be tradeoffs between precision (e.g., a numerical probability) and ease of acceptance by lay audiences (e.g., a listing status).

Information on designated status as non-threatened, threatened, vulnerable, endangered, etc., are available for many species, such that these designations can serve as a practical and operational approach to reflecting changes in extinction probability. The International Union for Conservation of Nature (IUCN), Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), U.S. Fish and Wildlife Service (FWS), and other government and non-governmental organizations have developed classification systems to reflect various degrees of extinction risk. An operational advantage of relying on classification systems such as these is that they are global in scope and updated regularly. Each classifies thousands of species and allows for reclassification based on new scientific information. Each relies on documented definitions and science-based guidelines for their designations. A disadvantage, at least in some cases, is that listing status might be subject to factors unrelated to extinction probability—for example the funding available to assess certain species or potential political pressure on listing agencies. When species listings are heavily influenced by non-biophysical factors such as these, the link to well-defined biophysical properties becomes attenuated

Some other studies have used habitats required to prevent extinction as the commodities being valued: protection of water flows "needed to protect an endangered species" (Berrens et al. 1996) and "required to reverse decline in habitat essential for the survival of the species" (Cummings et al. 1994); "critical habitat likely to avoid extinction" (Loomis and Ekstrand 1997); and "habitat [that] supports an endangered species" (Bauer et al. 2004). Although such indicators are common in past work, they violate a core property of linking indicators identified above (i.e., people must care about linking indicators or outcomes for their own sake, not just as drivers of, or proxies for, outcomes that directly matter). In cases such as these, it is not clear what is being communicated and valued—the underlying ecosystem change or the causally related change in species protection (Johnston et al. 2017c). Due to this ambiguity, "compound indicators" of this type are not ideal linking indicators for taxa.

In summary, there is robust evidence of existence value related to binary measures of taxa or species existence, both implied and explicit. These indicators can be presented in either deterministic or probabilistic terms. However, many of the descriptions in the past stated preference literature lack the properties of well-defined indicators described by Schultz et al. (2012) and therefore are problematic as linking indicators. To achieve linking-indicator status, this review stresses the importance of indicators being defined and measured in quantitative terms rooted in biophysical understanding.

B. Indicators of Abundance

We now turn to indicators of abundance unrelated to extinction risk. These indicators can be important because changes in organism abundance (e.g., population size) that influence extinction probability may also be valued directly and apart from any relationship to extinction—e.g., because people simply prefer having a larger population to a smaller population, *ceteris paribus*.[41] When addressing this type of indicator, it is important to distinguish between studies measuring use values (such as those that arise from commercial hunting or fishing, or recreational activities dependent on species populations) and those focused primarily on nonuse values (where existence value is a key component). Use value studies often focus on species that are relatively abundant. Changes in abundance (increases or decreases in population) are a common focus of such studies, but extinction or changes in extinction probability are not (for stated preference examples see Loomis and Larson (1994), Layton et al. (1999), and Bell et al. (2003). Here, we focus on cases in which WTP dominated by nonuse or existence values is influenced by changes in abundance.

The literature provides many such examples.[42] Although these studies often consider species that are uncommon or rare, a distinguishing feature of these indicators is that they do not emphasize extinction events or probability. These indicators can be presented either in cardinal form or as relative proportions, e.g., percentage or proportional changes in the abundance of marsh birds (Johnston et al. 2005b, Interis and Petrolia 2016), migrating fish (Zhao et al. 2013), or wetland dependent species compared to “historic” conditions (Milon and Scrogin 2006).

[41] Note that there are cases in which people might prefer smaller populations as well, reflecting negative existence values or cases in which value may not increase beyond certain levels. In any event, independent of the sign of the relationship between abundance and value, abundance may still be a linking indicator.

[42] Examples include stated preference studies of mountain caribou populations (Adamowicz et al. 1998), the number of fish (river herring and alewife) migrating in a watershed (Zhao et al. 2013), the number of naturally occurring Kakabeak shrubs existing in New Zealand forests (Yao et al. 2014), the population size of endemic dwarf buffalo (Barkmann et al. 2008), changes in the population of different salt marsh taxa (Johnston et al. 2005b), percentage changes in the population of predatory birds such as hen harriers (Hanley et al. 2010), and changes in the population size of general wildlife in the specific habitat (Nielsen et al. 2016).

Like extinction probabilities, other information may be represented relative to established thresholds, such as the number of fish-dependent wildlife species that are considered common within the affected area (Johnston et al. (2013;2017c). Finally, an implicit spatial dimension may be considered by describing abundance per unit of spatial area—or density. Additional discussion of spatial dimensions is provided in the following section. As above, the underlying biophysical properties of these indicators also determine whether they can be considered linking indicators. For example, qualitative descriptors such as "healthy" or "high" populations cannot be considered linking indicators, unless these descriptors are defined formally in terms of explicit, and quantitative, underlying measures (Johnston et al. 2012).

C. Spatial Dimensions: Global vs Local Existence Values

Implicit or explicit in many of the indicators discussed above is a spatial dimension—over what area was the indicator measured or relevant? Spatial dimensions are relevant to values for many types of ecosystem service changes, including those related to nonuse values for taxa. Glenk et al. (2020) review the literature on spatial dimensions of stated preference valuation, including spatial dimensions of nonuse or existence values. This review concludes that values "often depend on spatial aspects of the policy outcomes subject to valuation [and the] the respondents whose values are elicited."

The literature provides extensive support for the idea that existence values for taxa are conditional on their location and spatial relationship to respondents. For example, US residents were shown to have distinct nonuse WTP for the existence of Chinook salmon subspecies in two different regions where they currently exist (Johnston et al. 2015). Interis and Petrolia (2016) demonstrate that WTP for changes in wading bird populations is a function of their location.

Stevens et al. (1991) show that New England residents hold large existence values for the regional re-introduction of species that are common elsewhere. Morrison and Bennett (2004) find that use and nonuse values for otherwise identical increases in the number of bird and fish species present vary across different river catchments. Stated preference results in US EPA (2014b) and Johnston et al. (2013) show that nonuse WTP related to fish mortality reductions varies depending on the region of the US in which the changes occur and are valued.

An implication of these and other studies is that linking indicators for taxa should in some cases be defined relative to local and regional – not just global – existence, extinction risk, and abundance.

D. Summary Recommendations: Taxa Existence-Value Indicators

Robust evidence in the literature supports the hypothesis that measures of (1) existence or existence probability, and (2) abundance can serve as effective linking indicators for taxa. These measures may be presented for different taxonomic groupings or categories (e.g., one or more species), depending on the goals of analysis. They may be presented with explicit reference to thresholds or listing status (e.g., endangered, threatened) or as explicit quantities (e.g., 1000 wading birds, 20% probability of extinction). Where relevant, it can often be informative to communicate indicators in both cardinal and relative terms (e.g., 1000 wading birds is 1% of the estimated population in a particular area), as this can aid comprehension by non-experts (Johnston et al. 2012, Johnston and Zawojska 2020).

Not all indicators within these two categories qualify as linking indicators; other properties of ideal linking indicators for existence values (identified above) must also apply. For example, the indicator should be understandable to the target population, e.g., what does a listing status of "threatened" imply for the taxon in question? Indicators should also be presented with explicit spatial and/or temporal dimensions. For example, what are the spatial and temporal boundaries that define the indicators being presented (e.g., Endangered where? Abundance within what area? Over what time period?)? Quantitatively ambiguous or purely narrative commodity definitions should be avoided, such as "a large number of species" or unspecified "increases in" abundance or extinction probability. Measures such as these lack the precision to understand the quantities in question and encourage speculation (Johnston et al. 2012, Boyd et al. 2016) and do not lead to unambiguous biophysical representation.

Data availability may constrain the use of certain otherwise ideal linking indicators, particularly for spatially extensive applications where consistent measures are required. Hence, tradeoffs may be required. As an illustration, consider data related to binary existence versus non-existence (extinction). One might prefer a measure of extinction probability (or population viability probabilities for particular areas). However, such measures are often unavailable, requiring the use of less precise alternatives. The most systematic, global reporting of extinctions is conducted by the IUCN, which classifies extinctions in a variety of ways, including extinct, extinct in the wild (where there are individuals in captivity), and possibly extinct (where, despite gaps in recent observation or detection, rediscovery cannot be ruled out (Smith 1953, Meijaard and Nijman 2014). Within these classifications IUCN also tracks extinctions by taxon (e.g., mammals, amphibians, birds, insects, corals) and extinctions are dated. Accordingly, IUCN data is a practical, operationally convenient source of existence value-relevant biophysical metrics to relatively easy to track species-specific extinction events over time. The US FWS maintains similar information on an Environmental Conservation Online System (https://ecos.fws.gov/ecp/, accessed February 27, 2022). Data sources such as these can provide useful information for the development of various linking indicators related to the conservation status of taxa in different worldwide regions (including extinction).

Several caveats are worth noting. Estimates of current extinctions hinges on observations of the species in the past—this is a significant issue since it is estimated that millions of eukaryote species have not been catalogued and described (May 2010, Mora et al. 2011).[43] Thus there is corresponding bias in information toward species likely to be monitored and observed (perhaps towards larger organisms with larger ranges). The extinction of species endemic to locations remote from settlement and scientific observation, with very local ranges, and those difficult to detect due to size or behavioral traits (such as deep-water marine species) are less likely to be captured by existing data sources.

Now consider estimates of extinction risk, rather than extinction events. The science of extinction risk is complex as it builds on uncertain information about the life history of individual populations, their interactions with other taxa, and the dependence of those factors on habitat (in the broadest sense) and how those interdependent factors may change over the period of analysis. This literature grapples with significant data limitations, features diverse modeling and simulation approaches (Morris et al. 1999, Beissinger and McCullough 2002, Morris and Doak 2002), and wrestles with how to represent the certainty of the resulting estimates. Unfortunately, for our purposes the literature cannot be called on to deliver consistent, widely-available and regularly updated “change in extinction risk” measures for specific species. Although sufficient information may be available to develop reliable extinction-probability linking indicators for specific taxa in specific regions, this information is sporadic.

However, measures and data of this type do underpin national and global assessments

[43] We note in this regard the open question about whether people may hold an existence value for species they do not even know exist or that taxonomists have not described and catalogued.

designed to classify species according to their conservation status, as discussed above. FWS and IUCN designations of threatened species are widely available but lower resolution than explicit and perhaps localized probabilities of continued existence. The availability of information on underlying taxa abundance also varies, again particularly so if one considers specific spatial areas. Hence, those choosing linking indicators for particular applications may be compelled to choose lower resolution (or otherwise less ideal) measures when data are not available to support preferred options. Measures of this type have often been used when seeking to elicit nonuse values in the stated preference literature, suggesting their potential utility as linking indicators.

Data on the abundance of different species or taxa in specific regions is likewise sporadic. Distribution maps are available for many species, from a variety of sources (e.g. IUCN 2021, NatureServe 2021, USDA 2021). Estimates of species population sizes are available sporadically for some taxa, including species of concern (e.g., endangered species), game species, and some charismatic species over geopolitical areas such as US states or nationwide (e.g. He et al. 2000, Flather et al. 2013, Hanberry and Hanberry 2020, Cornell Lab of Ornithology 2021, USDA Forest Service 2021). Data sources and quality for these estimates are uneven, and data are frequently unavailable for smaller regions. Coverage of these data over different species and taxa also varies. As a result, past studies seeking to elicit values related to species abundance have relied on a wide array of often-inconsistent indicators for which data are available at the targeted scales. Cases such as these illustrate that while linking indicators on taxa abundance can often be developed for specific applications and regions, there is insufficient data to quantify these indicators broadly and consistently for a comprehensive set of taxa and across different spatial scales.

In concluding this section, we note that reference conditions (i.e., the baseline against which indicator levels are compared) are relevant to many of these indicators. For example, when presenting taxa abundance numbers for purposes of eliciting values or informing decisions, to what baselines should they be compared (e.g., historical abundance in the current location, the highest abundance currently observed elsewhere)? Multiple questions arise when seeking to determine the most relevant reference condition for specific types of indicators (Stoddard et al. 2006, Ode et al. 2016). Because questions related to reference conditions apply to nearly all types of linking indicators, these issues are discussed in a stand-alone section below.

6. Ecological Landscapes—Evidence and Proposed Linking Indicators

This section considers existence-value linking indicators for ecological landscapes, as defined earlier. An example is existence value related to an iconic ecological landscape like the Great Barrier Reef (Rolfe and Windle 2012). However, people may also place existence value on less iconic, locally significant landscapes. With existence values in mind, how should conditions and changes in ecological landscapes be represented in biophysical terms? In particular, how can they be represented in ways that both pass scientific muster and are meaningful to lay audiences (e.g., communicate constructs that people care about in understandable terms)?

Among the key challenges for linking-indicator development in this case is the *composite nature* of ecological landscapes. To reflect this composite nature, we emphasize indicators that—individually or jointly—reflect ecological landscapes as holistic systems, rather than characterizing narrow components of those systems (e.g., whether a species is present). In addition, we also largely set aside the related but distinct issue of how to define the geographic boundaries of these systems. Before doing so, however, we briefly review alternative approaches to defining geographic boundaries, in part to underscore that no one approach is clearly best for all applications.

A. Geographic Boundaries

Whereas taxa can be defined in purely biological terms, the boundaries of ecosystems or landscapes are an ecological and social construction. We could use purely physical definitions and associated boundaries to define specific rivers, lakes, glaciers, canyons, or mountains. We could define ecosystems and landscapes technically, and from a natural science perspective, as distinct combinations of physical and biotic characteristics.[44] We could use legal and institutional definitions, such as the boundaries of national parks or wilderness areas. Or we could rely on public perceptions of what constitutes an important ecosystem or landscape.[45] Any of these alternatives is potentially justifiable.

Even for a particular alternative there is no single, universally accepted approach to boundary definition. Consider the concept of an "ecosystem," which is heavily rooted in natural science-based notions of biophysical assemblage, processes, and interaction. Ecologists define an ecosystem as "any geographic area that includes all of the organisms and nonliving parts of their physical environment. An ecosystem can be a natural wilderness area, a suburban lake or forest, or a heavily used area such as a city." (Ecological Society of America 2021). As an

[44] This is akin to the definition of "biomes," which are distinct biological communities formed and supported by a shared physical environment (e.g., temperature, precipitation, soil characteristics, elevation).

[45] As an illustrative example of this ambiguity, consider Nebraska's Sand Hills in the US. This landscape has been defined ecologically (as a distinct ecoregion) and institutionally (as a national landmark). Not surprisingly, the two definitions do not yield the same geographic definition. We might also speculate that there are different cultural and psychological definitions of how lay audiences think of the "Sand Hills." For example, they are the "viewshed associated with travel along Scenic Byway Highway 2" or simply "hilly unfarmed parts of Central Nebraska."

illustration, linking indicators for the ecological condition of a particular river or watershed (beyond specific taxa) could be included in this category. Within academic ecology, the debate over the geographic definition and scale of ecosystems is vigorous and longstanding (Chapin et al. 2011, Schultz et al. 2012). For practical purposes, however, there is no single, clear *operational* definition of "an ecosystem" (Sagoff 2003).

Like ecosystems, landscapes can be difficult to bound in unambiguous terms. For ecologists one of the seminal papers on the topic starts with: "Landscapes surround us, yet curiously it is hard to find people with the same definition of a landscape" (Forman and Godron 1981). The boundaries of the Grand Canyon can be defined topologically and geologically or institutionally (i.e., based on the National Park's boundary). Depending on which dictionary is consulted, landscapes are "all the visible features of an area of countryside or land, often considered in terms of their aesthetic appeal," "the landforms of a region in the aggregate," "the visible features of an area of land, its landforms, and how they integrate with natural or man-made features," or "a large area of countryside, usually one without many buildings or other things that are not natural."[46] As with the definition of ecosystems, these definitions all rely on geographic concepts (an "area," a "large area," a "region") that are ambiguous. For these reasons, we set aside the issue of geographic boundary definition to focus on the properties of desirable existence value-relevant linking indicators, irrespective of a geographic boundary.[47]

[46] Lexico.com, Merriam-Webster.com, Wikipedia.com, and Diction.Cambridge.org, respectively.

[47] Geographic boundaries can be of primary importance for social and economic analysis, such as establishing the geographic domain considered for a benefit-cost analysis. However, in such cases, analyses typically proceed conditional on geopolitical boundaries established ex ante by the researchers for purposes of each analysis. These analytic boundaries and domains are typically independent of considerations on how one might define the geographical boundaries of an "ecosystem" or "landscape" in conceptual terms. Equally, the importance of specifying operational boundaries for specification of linking indicators is well discussed in Ringold et al. (2013) and US Environmental Protection Agency (2020).

B. Ecological Landscapes as Bundles of Features: Implications for Linking Indicators

Among the primary challenges for ecological-landscape commodity definition is that these systems are characterized by bundles of biotic and physical features, rather than one defining feature. They are typically assemblages of plants and animals co-located with physical terrain such as estuaries, rivers, canyons, mountains, hills, and plains. What is valued about these systems may vary. The Grand Canyon may be valued primarily due to abiotic geological or hydrological features. In other cases, only (or primarily) biotic elements may be relevant, such as biodiversity or species richness within rainforest systems. An additional complication is that the quality of the landscape's individual features may also be important in some instances (e.g., the clarity of Lake Tahoe's water).

Because ecological landscapes are typically valued as bundles of biotic and/or abiotic conditions, compound biophysical measures are often required to capture the portfolio of features and qualities that give rise to existence value. Accordingly, we should expect linking landscape indicators to themselves have a compound nature – either as portfolios of individual indicators or as indices that synthesize multiple forms of information into a single metric. The fact that relevant linking indicators can vary across different types of ecological landscapes presents a challenge to the development of a concise set of recommendations for these measures.

Reflecting this challenge, one encounters a vast array of different approaches to communicating ecosystem and landscape status in the valuation literature and elsewhere—not all of them well-defined from a biophysical perspective. The uneven approach to characterizing ecosystem status and condition in the economics literature (e.g., on valuation, ecosystem services, etc.) leads to challenges interpreting, comparing, and generalizing the results of this work. Sometimes status is conveyed in the literature using multiple independent indicators communicated as a group, whereas in others it may be communicated using an indicator that broadly integrates the condition of an ecosystem or landscape, e.g., measures of biotic integrity. In many other cases—such as valuation studies designed to prioritize conservation areas—a single, non-composite measure is used (e.g., species richness), despite acknowledgement that these individual measures provide only a partial representation of ecosystem properties relevant to economic value (Dissanayake and Ando 2014).

Methods to quantify biotic integrity as a general concept within various disciplinary literatures vary from the use of formal multi-metric indicators such as Indexes of Biotic Integrity (IBIs) (e.g. Stoddard 2008, Johnston et al. 2011, Holland and Johnston 2017)[48] , O/E (observed-to-expected taxonomic composition; Moss et al. 1987, Hawkins et al. 2000), or biological condition gradients (or BCGs; Davies and Jackson 2006, U.S. Environmental Protection Agency 2016, Lupi et al. 2020), to the use of narrative or descriptive measures that do not meet the criteria for linking indicators (e.g., see discussions in Johnston and Russell 2011, Schultz et al. 2012, Boyd et al. 2016, Olander et al. 2018). The valuation literature often sidesteps the challenge of developing quantitative and well-defined indicators by using poorly defined narrative descriptions such as the presence of "many animal species," "high biodiversity," or areas in "good health." As described by Johnston et al. (2012) and Schultz et al. (2012), narrative descriptions such as these are unlikely to convey an accurate understanding of ecosystem condition amongst experts or the lay public, unless paired with quantitative information that defines each presented category. In addition, measures of this type are poorly suited for quantitative integrated modeling, because imprecisely defined narrative or indicators cannot be

[48] There is a mature ecological literature devoted to multi-metric indicators of this type, including debates over their validity and salience (e.g. Karr 1981, 1991, Suter 1993, Stoddard et al. 2008a, Stoddard et al. 2008b, Ruaro and Gubiani 2013, Wurtzebach and Schultz 2016, Carter et al. 2019, Ruaro et al. 2020, Karr et al. 2021).

linked unambiguously to biophysical models (see discussion in Lupi et al. 2020). Even the use of categorical indicators which may be used in biophysical models or in a linked set of models can reduce the certainty or sensitivity of a quantitative analysis.

Patterns such as these in the economic valuation literature speak to the challenge of ecological landscape commodity definition for existence values, particularly given the heterogeneity of conditions that may be valued. For example, the question of what constitutes degradation and alteration of ecological landscapes emerges clearly as an issue – as do associated concepts like what is ecologically "healthy," "intact," and "natural." The latter question relates, in part, to appropriate reference conditions for these systems. Moreover, in many cases laypeople may not have a clear, well-defined prior understanding of exactly what biophysical measures characterize the "things that they value." Unpublished data from dozens of focus groups conducted by the authors suggest that non-experts will often describe existence values for ecological landscapes in terms of general concepts such as "the way it used to be," "undisturbed," "natural," or a functioning "web of life" where "everything is connected." Ideally, well-defined linking indicators could represent these public conceptions, but by themselves these ideas are too vague to be useful.

Again, we direct our review primarily to the stated preference literature and other studies focusing primarily on existence value–as opposed to use values.[49]

C. Categories of Ecological Landscape Indicators

Challenges such as these notwithstanding, we posit that there is sufficient evidence in the

[49] Valuation studies based on use value related commodities often find a correlation between use and nonuse values, as shown by the meta-analysis of Johnston et al. (2003). This suggests that use value linking indicators may also be relevant indicators for nonuse values. Moreover, many past studies have shown distinct use and nonuse values for the same biophysical measures or outcomes (e.g. Hanley et al. 2003, Loomis 2012).

literature to support guidance regarding generally applicable linking indicators for ecological landscape existence values. Based on a review of this literature, we propose three general categories of potential existence-value linking indicators. The prospective relevance of each proposed indicator category is supported—in general terms—by the prior valuation literature. However, as revealed by the discussion that follows, we also find that many areas of the literature apply measures that lack the properties required for linking indicators.

The first category relates to the *areal extent* of a particular ecological landscape. These indicators are analogous to the abundance measures associated with taxa. They describe the spatial quantity (abundance) of an ecological landscape globally, nationally, regionally, or locally. In the extreme, major losses in areal extent – globally or locally – can be metaphorically viewed as extinction events: in this case not the extinction of a species, but the extinction of a particular kind of ecological landscape. Measures of this type can also be extended to accommodate quality levels or losses due to major degradation or events—such as acres of a temperate coniferous forests lost due to forest fire in the Western US. Such measures answer the simple question, "how much of the ecological landscape exists," and are commonly linked to existence values in the literature.

The second category of ecological landscape indicators relates to *species diversity and composition* within a system, for example reflecting species richness or the prevalence of invasive versus native species (Jacobsen et al. 2008; Dissanayake and Ando 2014). We differentiate these measures from linking indicators for individual taxa because they reflect composite, heterogenous assemblages of taxa. Moreover, they are typically used within economic and social analysis, not as a means to characterize the underlying individual taxa (whether one or more species exists), but rather as a means to characterize the ecosystem itself, e.g., the diversity of life it supports.

The third category includes *measures of holistic or emergent ecosystem condition* such as indices of biotic integrity (IBIs), O/E, or Biological Condition Gradient classifications which each serve as an indicator of biological integrity. Indices of this type have been shown by Johnston et al. (2011) – using the example of an IBI – to be capable of capturing existence values for holistic or emergent ecosystem properties not otherwise captured by single-metric indicators. We also include in this category measures of *environmental quality, or other components of ecological integrity* viewed as key to people's sense of the health, integrity, and naturalness of ecological landscapes. An example of a physical integrity measure is instream flows within rivers (Loomis 2012) relative to a reference state (e.g. Kaufmann et al, in preparation). An example of a chemical integrity measure is water quality (Johnston et al. 2005a, Johnston et al. 2017a, Johnston et al. 2019) relative to a reference state (Herlihy and Paulsen 2022). We further include environmental quality measures (such as air quality and visibility in scenic national parks) that help to define a landscape's defining "sense of place," even though they may not be directly connected to ecological health, integrity, and naturalness.

As above, we emphasize that these are not the only possible linking indicators that can be developed for ecological landscapes. For example, it is possible to characterize these systems using collections of individual measures that are applicable to each system. However, collections of features such as these are often idiosyncratic to specific systems and difficult to generalize. Here, in contrast, we seek to characterize a small number of more holistic linking indicators likely to apply to existence values within a wide range of ecological landscapes.

i. Areal Extent Indicators

Among the quantitative indicators commonly associated with nonuse values in the economic literature are measures of areal ecosystem extent. The existence of greater areal extent is associated with higher existence value estimates. Measures of this type, of course, presume that one can identify the boundaries of the "ecosystem" under study such that its extent can be determined. As noted above, identification of ecological landscape boundaries can be difficult. However, predetermined boundaries can facilitate this task for certain preselected ecological landscape types (e.g., arboreal forest, salt marsh, freshwater wetland, coral reef). For example, government agency data and increasingly available GIS data layers (e.g., provided by government agencies, university researchers, NGOs and others) often provide boundaries for specified ecosystem types based on predetermined criteria or classification systems. Linking indicators for valuation or other purposes can then be determined conditional upon these data and boundaries (cf. Bateman et al. 2002, Johnston and Ramachandran 2014, Bateman et al. 2016, Johnston et al. 2016, Sagebiel et al. 2017; Glenk and Martin-Ortega 2018, Badura et al. 2020).

For example, the literature has estimated nonuse values associated with the areal extent of systems such as wetland or marsh (Johnston et al. 2002b, Lupi et al. 2002, Morrison et al. 2002, Bauer et al. 2004; Johnston et al. 2005b, Petrolia et al. 2014, Johnston et al. 2018), beaches (Johnston et al. 2018), forest or wilderness areas (Adamowicz et al. 1998, Czajkowski et al. 2017; Sagebiel et al. 2017; Varela et al. 2018), natural riparian vegetation (Holland and Johnston 2017, Uggeldahl and Olsen 2019), Danish heath (Jacobsen et al. 2008), rainforest (Rolfe et al. 2002; Siikamaki et al. 2019), coral reef (Rolfe and Windle 2012, Rogers 2013, Aanesen et al. 2015), peatland (Glenk and Martin-Ortega 2018) sea- or eelgrass (Johnston et al. 2002a, Rolfe et al. 2002, Rogers 2013), grassland (Sagebiel et al. 2017), and migratory fish habitat (Johnston et al. 2011, Johnston et al. 2012), among many others. This literature also provides evidence that people often hold values for changes in the areal extent of these systems, above and beyond the related value of the taxa and ecosystems within those areas.[50] Similar indicators that reflect the probability of future changes in areal extent are also likely to serve as useful linking indicators in some cases.

Some of the above measures confound biophysical existence with degrees of protection (e.g. Bateman et al. 2009, Czajkowski et al. 2017, Spencer-Cotton et al. 2018). Moreover, the existence value per unit area may not only depend on the binary existence (versus lack of existence) of an ecosystem type, but also on its quality or other characteristics. This has led some studies to qualify the definition of ecological landscape. Examples include narrative descriptors such as “good quality” peatland (Glenk and Martin-Ortega 2018), coral reef in “good health” (Rolfe and Windle 2012), or the proportion of river areas with “healthy” vegetation (Morrison and Bennett 2004). In other instances, characteristics are illustrated via stylized graphics—for example images showing the presence or absence of a forest shrub layer (Varela et al. 2018). Qualifiers such as these can be defined in terms of quantifiable underlying ecosystem properties, but often lack concrete underlying quantitative definitions (at least when presented to non-experts). More broadly, terms such as “good quality” and “healthy” are less than ideal because (absent additional information) they can be interpreted in different ways. For example, the spatial area of “good quality peatland” would only qualify as a linking indicator only if “good quality” is defined in a quantifiable and meaningful way with known certainty.

Taken to an extreme, quality-related issues such as these raise questions concerning what

[50] Choice experiments such as those cited in the above paragraph often include independent attributes communicating geographical size (e.g., of an ecosystem or preserved area) and the characteristics of taxa or ecosystems within that areal extent. Analyses often reveal independent values for both types of attributes. Results such as these suggest that individuals are often willing to pay to expand the geographical size of an ecosystem or preserved area, even if the number/type of organisms (or other ecosystem properties) in that area remain constant.

it means for an ecosystem type to "exist" in a particular area. Given our emphasis on extinction events and extinction probabilities in the discussion of species, it is worth reflecting on whether the concept of extinction could be applied to landscapes as well. Can a landscape go extinct? In the case of extreme degradation and alteration, yes—an example is coral reef that is entirely dead or bleached, such that no living coral remains within the area of interest. Accordingly, one strategy is to quantify and track such unambiguous "landscape extinction events" or changes in their probability, by focusing on natural resource changes associated with fundamental land use change (e.g., felling or burning a rainforest to be replaced with row crops), converting a wetland to an urban area, allowing an abandoned farm to revert to woodlands, etc. If a landscape whose existence is valued disappears entirely as a landscape, or if it is threatened with destruction, those events and probabilities are directly relevant to welfare.

We note, however, that the definition of what constitutes "extreme" degradation and alteration can be subjective. For example, does a coral reef that is 99% bleached continue to "exist" as an ecological landscape? Would Lake Tahoe continue to exist as an iconic landscape if the lake were to become heavily eutrophic with near-zero water clarity? A related point is that because landscapes are bundles of features and qualities, it is not clear that the removal of a single feature, or degradation of a single quality would constitute an extinction in the public's eyes. Would the Grand Canyon landscape be "destroyed" if the Colorado River no longer flowed through it, or if the river became significantly more polluted? These changes could be considered extreme forms of degradation and alteration but nevertheless not constitute – to the public – the "extinction" of the Grand Canyon. However, they could very well influence existence values. To a large extent the existing valuation literature sidesteps these questions and presents areal extent measures (sometimes along with associated quality measures or characteristics) as unambiguous constructs.

In summary, areal extent measures can serve as defensible linking indicators, subject to important qualifications. The type of ecological landscape under consideration must be clearly defined, quantifiable, and understandable. Identifying ecological landscapes using poorly defined narrative descriptors such as "good quality" (without underlying details to define such categories) will lead to poorly defined measures—even if the spatial area itself (e.g., 10,000 hectares) seems superficially well-defined and readily understood.

ii. Measures of Species Diversity and Composition

Another type of indicator commonly linked to nonuse values in the literature includes measures of species diversity and composition. These are often framed in terms of biodiversity, species richness, or native versus invasive species distributions (e.g., Jacobsen et al. 2008, Meyerhoff et al. 2009, Johnston et al. 2012; Dissanayake and Ando 2014, Varela et al. 2018). We distinguish these from the taxa linking indicators described earlier because of the degree to which such measures explicitly reflect the joint existence, abundance, or distribution of *multiple* species simultaneously. Hence, these indicators are not designed to communicate whether a species itself exists, but instead use species diversity and composition measures to characterize a broader system.

The literature in this area is extensive and diverse, both in terms of applications and in terms of the types of measures used. While many studies use indicators that are well-defined from a biophysical perspective, others use narrative or otherwise ambiguous biophysical descriptions. Examples of taxa composition measures in the stated preference literature include "the number of fish species in good health" (Rolfe and Windle 2012), the presence of "special" plants and animals (Rolfe et al. 2002), "medium" or "high" wildlife or plant and animal diversity (Meyerhoff et al. 2009, Newell and Swallow 2013), the number of red-listed species that will be preserved (Jacobsen et al. 2008), the percentage of rainforest species "threatened with extinction" (Siikamäki et al. 2019), narrative or graphical descriptions of biodiversity (e.g. Carlsson et al. 2003, Birol et al. 2009, Jacobsen et al. 2011; Glenk and Martin-Ortega 2018)[51,52], the number of native species present (Morrison and Bennett 2004); endangered or protected species present (Morrison et al. 2002); the type of fish species present, e.g., Salmon, trout and course fish versus only course fish (Hanley et al. 2006), the number of "fish-eating species that are common … such as egrets, osprey, otters, eagles, turtles and mink" (Johnston et al. 2011, Johnston et al. 2012), or descriptions of whether "rare" and "common" species decline (Christie et al. 2006).

Although many studies do not provide explicit quantitative measures of composition, richness or diversity, there are exceptions. For example, Nordén et al. (2017) report statistically significant values associated with variations in tree species compositions, reported in terms such as "33% spruce, 33% pine, and 33% hardwood" versus "100% spruce." Similar forest taxa distributions (e.g., percent broadleaf, conifers, etc.) are used by De Valck et al. (2014). In a third forest example, Horne et al. (2005) uses a 100-point index of tree species richness, "derived using the number of tree species and the amount of decayed wood on the ground." Filyushkina et al. (2017) characterize valuation scenarios using categorical measures of various structural aspects of forest stand diversity, including the proportion of stands that have the same species

[51] As an example, Jacobsen et al. (2011) describes a particular biodiversity level ("some diversity") as a situation in which "Many animals [are] distributed among a larger number of ordinary species, including small birds. Vulnerable and rare plants are only threatened by extinction in a few places."

[52] Birol et al. (2009) describe "low" versus "high" biodiversity narratively in terms of the "Number of different species of plants and animals, their population levels, number of different habitats and their size in the wetland ecosystem in the next 10 years."

and height structure. In one of few studies using a formal biodiversity index, Weller and Elsasser (2018) elicit WTP using (among other attributes) an index of biodiversity that "measures species diversity and landscape quality by monitoring the population development of characteristic bird species; the sub-indicator for forests embraces eleven bird species." Varela et al. (2018) quantify species richness within categories such as "herbaceous species," "butterfly species," and "bird species." Dissanayake and Ando (2014) characterize the outcomes of grassland restoration using measures that include the number of bird species present, density of birds per acre, and number of endangered bird species present.

Even a superficial review of this extensive literature suggests that: (1) measures of taxa composition and diversity—even if biophysically ambiguous—are often relevant to the public, (2) there is no standardized approach to quantifying and communicating biodiversity or taxa composition for valuation purposes, (3) many applications rely on narrative, pictorial or categorical descriptors without clearly stated quantitative foundations, and (4) few applications use measures that meet all our linking indicator criteria. Even studies that quantify species richness (i.e., the number of species of particular types within a system) often qualify these numbers with seemingly ambiguous descriptors such as "special" or "rare" that may be interpreted differently by the lay public (and/or by experts), and hence add ambiguity to the measure.

Interestingly, despite numerous formal biodiversity measures within the biological and ecological literatures such as the Simpson's Index (e.g. Pavoine and Bonsall 2011, McGill et al. 2015), these measures are rarely used to communicate biodiversity within valuation studies or the social sciences more broadly. Simpler measures, such as species richness are more common. Moreover, there is lack of consensus even within the biophysical literature on the metrics that should be adopted to measure and communicate biodiversity change (Brummitt et al. 2017, Turak et al. 2017). It has been argued that the stated preference literature “focus[es] mostly on individual species and habitats but do[es] not value the diversity itself” (Laurila-Pant et al. 2015). This may be due to the relative complexity and opaqueness of these measures. Without information on the extent to which these measures can be understood as meaningful by non-experts, it is unclear whether they could serve as effective linking indicators. The avoidance of these measures within the valuation and broader social science literature (with the exception of rare studes such as Weller and Elsasser 2018) suggests that classical ecological diversity indices may not always serve as useful linking indicators—despite their familiarity to those in the biophysical science community.

Alternative compound measures to quantify the “value” of biodiversity from a socio-ecological perspective have been proposed (Laurila-Pant et al. 2015). However, these measures commonly reflect ad hoc combinations of ecological and socio-economic information that confound biophysical data with expert assessments and can result in the same interpretability problems as conventional ecological indices. An additional challenge for linking indicator guidelines—highlighted by works such as Baumgärtner (2005) and Laurila-Pant et al. (2015)—is that the relevance of different diversity indices depends on the decision-making context. General works on biodiversity valuation such as Pearce and Moran (1994) discuss the need for consensus indices of biodiversity but are equivocal in their recommendations and stop short of explicit guidance.

We offer the similarly equivocal conclusion that (1) there is evidence that taxa composition and diversity measures can serve as linking indicators for existence values associated with ecological landscapes, but (2) there is insufficient evidence from the literature to support a particular composition and diversity indicator. Despite numerous formal diversity and species composition measures available from the scientific literature, the lack of consensus around which are most ecologically appropriate, concern over any particular measure's generalizability, and the challenge of making such measures comprehensible to lay audiences prevents us from recommending any particular measure as a linking indicator. However, we can refer to the criteria for linking indicators described in Section 3. In particular, we advocate the use of indicators based on of specific, quantitative numbers (e.g., number of native species present), proportions (e.g., proportion of species that are native), or other well-defined quantitative measures that can be readily and clearly interpreted by the lay public. When qualifiers are added (e.g., "threatened"), the interpretation of these terms should be unambiguous (e.g., "threatened" as determined by what criteria and/or by whom?). More complex indicators (e.g., biodiversity indicators) may serve as linking indicators in some cases, but only if these measures can be designed and communicated in a way that allows for clear interpretation by the lay public. This is an area where more research is needed—particularly to reconcile the social science literature devoted to communicating and valuing biodiversity with the biophysical literature devoted to quantifying it.

iii. Composite Measures of Holistic Ecosystem Condition

As described by Johnston et al. (2011), the lay public will often express—in both quantitative and qualitative forms—preferences for "the overall condition or naturalness of an ecosystem", often considered "relative to an undisturbed referent." These preferences are distinct from values held for individual aspects of the system such as the existence or abundance of specified taxa. Reflecting a growing literature in this area, there is increasing consensus that emergent, holistic properties such as these can be directly relevant to the public, particularly when nonuse values are considered (Boyd et al. 2016). Here the valuation literature builds upon work in the ecological sciences on multi-metric indicators of ecosystem condition, such as indices of biotic integrity or IBIs (Karr 1981, Karr and Dudley 1981, Karr et al. 1986, Stoddard et al. 2008a, Stoddard et al. 2008b, Ruaro et al. 2020), and biological condition gradient classifications or BCGs (Davies and Jackson 2006, U.S. Environmental Protection Agency 2016, Gerritsen et al. 2017, Bradley et al. 2020, Lupi et al. 2020). Although the shortcomings and controversies associated with measures of this type are well known (Suter 1993, Suter 2001), including potential challenges related to interpretability, growing evidence from focus groups and formal valuation analyses in economics suggests that they may serve as valid holistic indicators of ecosystem condition. Similar single-metric indices of ecosystem condition include O/E metrics (Observed Number of Taxa/Expected Number of Taxa; (Moss et al. 1987, Hawkins et al. 2000)).[53]

This literature provides consistent evidence of nonuse values associated with multi-metric indices of ecosystem condition, typically applied to aquatic systems (Johnston et al. 2011, Johnston et al. 2012, Johnston et al. 2013, Zhao et al. 2013, Johnston and Ramachandran 2014, Johnston et al. 2016, Holland and Johnston 2017, Johnston et al. 2017c). In contrast to much of the valuation literature on taxa composition cited above, the stated preference surveys reported in this literature often provide information on the underlying structure of the applied indices. As described by Johnston et al. (2011), this provides a quantitative and "ecologically grounded

[53] Because O/E measures are derived as a ratio of observed-to-expected taxonomic composition, it is not always clear whether these reflect indicators of ecological landscape condition or indicators of the underlying taxa. Similar measures encountered in the valuation literature reflect the relative number of possible rare or native species that exist (or will continue to exist) in a given ecosystem. An example is Jacobsen et al. (2008), whose valuation scenarios include the measure, "How many of the 25 red-listed species on the heath will be preserved? (0, 5, 12 or 25 species)". However, this is communicated as a measure of "species preservation" rather than ecosystem condition.

means to quantify" the "ecological condition of [a] system, apart from values for other services delivered by that system", while avoiding the ambiguity associated with narrative and other descriptions common in the valuation literature (Schultz et al. 2012). In other words, participants are often (though not always) provided information about both the overall index and its component measures.

This literature also provides consistent evidence of nonuse values associated with multi-metric indices of water quality. Beginning with Johnston et al. (2003), there have been multiple meta-analyses demonstrating systematic nonuse values for quantifiable water quality changes in US lakes, rivers, and estuaries, measured across different studies, building on the large valuation literature in this area (cf. Johnston et al. 2005b, Van Houtven et al. 2007, Johnston et al. 2017a, Johnston et al. 2018). These changes have been measured using various metrics, most commonly a variant of a standard water quality index, or WQI (Abbasi and Abbasi 2012, Walsh and Wheeler 2013). Johnston and Bauer (2020) illustrate the calculation and use of WQIs of this type for valuation purposes (i.e., as linking indicators). Although the limitations of such measures are known (e.g. Walsh and Wheeler 2013), they have been repeatedly demonstrated as an effective means to predict both use and nonuse values for water quality change and are among the most widely used indicators in the valuation literature. Variants of these indices have been developed for specific contexts such as lake acidification and to accommodate visual dimensions of water quality (e.g. Bateman et al. 2005, Bateman et al. 2011).

In Europe, water quality valuation studies tend to be framed differently, in terms of areas or lengths of rivers at different categorical quality levels (e.g., "good condition"). However, the categories are grounded in similar underlying quality indices defined for consistency with the EU Water Framework Directive (Bateman et al. 2011, Kataria et al. 2012, Martin-Ortega et al. 2012, Jorgensen et al. 2013, Brouwer et al. 2016). In some cases, these categories are described in ways similar to indices of ecological condition, as described above. For an example, see Schaafsma et al. (2012). Hence, it can sometimes be difficult to disentangle measures of water quality itself (measured in biophysical units such as a formal WQI) from more general measures that communicate the effects of water quality on other valued outcomes (such as water color, clarity, and supported taxa).

Criticism of multi-metric index measures such as these include the lack of clear interpretability of measurement units and sensitivity to index construction and functional forms (Hill et al. In Press; Suter 2001, Walsh and Wheeler 2013). For example, some have argued that the units in which IBIs are measured may not be meaningful to the lay public (Olander et al. 2018). Moreover, there is a possibility that some members of the public might not value "overall ecological condition" directly, but rather might use this information to speculate about levels of other outcomes for which they have direct preferences, such as use-related visual or scenic attributes (Johnston et al. 2017c). In addition, despite the increasing prevalence of these indicators in the valuation literature—the number of studies using such measurements is still relatively small.

The role of reference conditions is particularly relevant when developing these composite measures. Reference conditions reflect the upper anchor point that determines the best outcome or score on whatever composite measure is considered (e.g., a score of 100 on a 100-point WQI or IBI). The diverse approaches to defining "best" are described below. Suffice it to say, the validity and interpretation of the measure is thereby conditional, first on a clear and common understanding of what that measure means, and second on both the biophysical and social validity of the reference conditions (Bouleau and Pont 2015). Although it is beyond the scope of this article to review the various methods that may be used to quantify reference conditions and the many questions that can arise when doing so, the **Reference Conditions** box below introduces the topic.

Reference Conditions

Throughout we note that "reference conditions" are relevant to the representation of biophysical features for existence values. Stoddard and his colleagues (2006) summarized four conceptual approaches to defining reference conditions. These are: Minimally Disturbed (MDC) which approximates the pristine condition; Historical Condition (HC) which describes ecosystem condition at a specified time in the past; Least Disturbed Condition (LDC) which describes the best that can be found currently within a region; and Best Attainable Condition (BAC) which is equivalent to the expected ecological condition of least-disturbed sites if the best possible management practices were in use for some period of time. Given the importance of the reference conditions there has been an extensive effort to quantify metrics that represent these states in both aquatic and more recently in terrestrial systems (Carter et al. 2019). While methods exist to quantify some metrics, at some times in the past, these methods are extremely limited in terms of the comprehensiveness of what they capture (though they can be narrowly useful in MDCs, and HCs)[1] or are based on expert judgment (Davies and Jackson 2006). Thus, for pragmatic reasons, the Least Disturbed Condition (LDC) reference state is most readily quantified for a broad range of metrics and most frequently used in ecological analyses. The question for defining and quantifying the biophysical features that represent existence values is which, if any, of these conceptual definitions match the reference conditions embedded in the understanding that people hold. The current valuation literature provides little insight on these questions.

[1] As for quantifying the BAC state, Stoddard (2006) describes it as "a somewhat theoretical condition predicted by the convergence of management goals, best available technology, prevailing use of the landscape, and public commitment to achieving environmental goals."

Also unclear is the relative suitability of alternative types of multi-metric or single metric indicators of ecosystem condition, such as O/E versus IBIs. Biophysical comparisons have been completed for BCGs versus IBIs (U.S. Environmental Protection Agency 2016), and some valuation work has applied variants of BCGs (Lupi et al. 2020). However, we are aware of only one recent work (Hill et al. In Press), that examines the suitability of O/E relative to an MMI with respect to the interpretation of each measure by focus group respondents. This study concludes that O/E was a more interpretable measure than an MMI.

iv. Other Defining Measures of Ecological Landscapes

The categories discussed above represent indicators designed to capture holistic existence features of ecological landscapes. However, other indicators may be relevant, including measures that characterize essential or defining characteristics of ecological landscapes as identified places. These characteristics must be as defined or perceived by the people whose values are being studied. This section presents some commonly applied indicators of this type.

The first examples relate to defining visual characteristics of landscapes—such as water clarity in Lake Tahoe or visibility (influenced by air quality) in iconic locations such as the Grand Canyon. Measures of this type can be interpreted as defining characteristics of these iconic places and are thus directly relevant to existence values.

First consider indicators of air quality, when considered as defining features of ecological landscapes.[54] Measures of visibility (Smith et al. 2005, Yoo et al. 2008, Rizzi 2014, Boyle et al. 2016) provide an obvious candidate for linking indicators of this type. Although visibility is

[54] In terms of *use* values, air quality primarily affects social welfare via impacts on human mortality and morbidity.

often represented using photographs in surveys, it is typically defined and categorized using quantitative measures such as $\mu g/m^3$ of $PM_{2.5}$, visual range (VR), deciviews (DV), or related measures (Malm 1999). Measures such as VR can also be communicated simply as visibility distance (e.g. km of visibility; Yoo et al. 2008). Long-term monitoring and datasets are available for many of these visibility measures, for areas such as national parks (Boyle et al. 2016). If measures such as these can be communicated in a way that is understood clearly by non-experts, they can serve as valid linking indicators for ecological landscapes within which visibility is a defining feature.

Measures of water clarity can serve a similar purpose in some types of aquatic systems, such as lakes and coral reefs. Various approaches have been used to communicate water clarity in stated preference studies, including simple measures such as Secchi depth (Johnston et al. 2017c). These measures have been shown to be directly relevant to both use and nonuse values (Poor et al. 2001, Kerr and Sharp 2008, Johnston et al. 2017c, Klemick et al. 2018), and hence are good candidates for linking indicators. For purposes of communication, measures such as Secchi depth are preferred linking indicators compared to more “opaque” measures such as turbidity, as the former is measured in units that are easily understood by the lay public (e.g., feet of visibility for a bright object). Turbidity and similar measures—while more precise—are quantified in units that are understood solely by experts, such as nephelometric turbidity units (O'Dell 1993). However, it is also worth considering that Secchi depth is often considered to be a poor indicator for some types of aquatic systems, such as estuaries (Pedersen et al. 2014) and is often not measured in streams. Hence, the utility of such measures as linking indicators depends on context.

Environmental *quantity* measures can also serve as defining characteristics for some types of ecological landscapes. An example is water quantity (such as instream flows or water levels) within rivers (Loomis 2012) or aquifers (Koundouri et al. 2014), although in some cases these measures are framed in terms of a minimum quantity required for species existence (Berrens et al. 1996). Where water quantity itself is a defining characteristic of a system, it can be measured in various ways, such as volume per unit time or depth (e.g., for reservoirs or groundwater). However, there are few examples of well-defined measures of this type in the valuation literature. Most surveys present quantity levels using general descriptive labels or narrative descriptions such as “increased” and “limited” (Koundouri et al. 2014) or confound quantity and quality measures (Tempesta and Vecchiato 2013, Damigos et al. 2017). Hence, there is little evidence to support guidelines regarding the most effective or relevant linking indicators for water quantity.

There are also multiple stated preference studies that present quantity or density measures related to visible landscape features such as forest trees as part of total WTP or preference estimation (including nonuse or existence values). These measures also vary widely across the literature. These studies are distinct from the “spatial area” studies above, in that they do not quantify ecosystem or landscape areas, but rather the density or quantity of features within those areas. For example, Edwards et al. (2012) evaluate public preferences over forest stands distinguished by structural measures such as the variation in tree spacing and cover within each stand. Giergiczny et al. (2015) present similar attributes both visually and using narrative categorical descriptions, focusing on both density and diversity measures. The specific measurements used to underpin these categorical and visual representations are not described. Variants on indicators of this type are found throughout the literature evaluating public WTP for changes in forest structure and management. Similar measures of visual features are found in stated preference studies that consider other ecological landscape types. For example, considering grasslands, the choice experiment of Dissanayake and Ando (2014) quantifies the "percentage of restored land area that will be covered by wildflowers," implicitly presuming that wildflower coverage may be categorized as either present or absent on any given area.

In summary, while various measures may serve as defining characteristics of ecological landscapes such as lakes, forests and grasslands, these measures are likely to be idiosyncratic across different systems. Some but not all of the measures used in the associated valuation literature have desirable properties of linking indicators, such as being defined in quantitative, repeatable and unambiguous terms. Visibility and clarity indicators, for example, are relevant to existence values for some systems, because they define how these systems are perceived and defined by the public. These indicators can also be presented as direct, quantitative and repeatable measures. Linking indicators of this type are likely to be relevant to some but not all ecological landscapes.

D. Summary Recommendations and Data Availability: Ecological Landscape Indicators

The wide diversity in metrics, measures, and descriptions used to characterize ecological landscapes in the valuation and public preference literatures complicates the search for a concise and universally applicable set of preferred linking indicators. Also, there may be a tradeoff between complex indicators preferred in the biophysical literature (e.g., for biodiversity) and simpler measures more appropriate to public communication. Given these challenges, we focus our summary recommendations for ecological landscape linking indicators on those measures (a) for which there is strong and consistent evidence of relevance across the literature, and (b) that are likely to be generally relevant, as opposed to salient in only a few specific contexts.

Regarding measures of areal extent, there is widespread use and acceptance of publicly available datasets that map land use and cover, vegetation type, ecosystem type and other attributes that may be used to define some types of ecological landscapes—from salt marshes to seagrass beds to coniferous forests. These measures are hence promising candidates for linking indicators, conditional on a clear and shared understanding of how the extent of each ecological landscape is defined.

Regarding measures of taxa composition and diversity, we note their variety and the difficulty of clearly communicating their meaning to the public (Schultz et al. 2012). Some areas of the literature (e.g., forest valuation) have shown a degree of consistency in the types of quantitative metrics that are used, e.g., taxa composition across forest stands and diversity in tree height. However, other areas of the literature (e.g., aquatic systems) are dominated by studies using narrative or descriptive indicators that lack consistency across applications. In addition, the measures used to communicate biodiversity in the social sciences (e.g., in surveys) rarely coincide with parallel measures used in the biophysical literature, although species richness measures are common across both areas. Given this lack of consistency, we feel there is insufficient evidence to identify a concise set of "best" existence value linking indicators for taxa composition and diversity. These measures are unquestionably relevant to existence values across many types of ecological landscapes, but additional research is needed on their role as linking indicators.

Another issue deserving additional research relates to composite measures of holistic ecosystem condition such as biotic integrity. As described above, biotic integrity and other holistic attributes can be quantified with multi-metric indices including formal multi-metric indicators such as IBIs or BCGs, or single-attribute indices such as the O/E metric. Yet there are relatively few valuation studies that use any of these quantities. Hence, despite growing evidence that holistic "ecosystem condition" is linked to existence values across multiple types of landscapes, the literature is not yet sufficiently developed to identify unequivocally superior linking indicators. Unlike many measures discussed in this section that are readily available, multi-metric indicators such as IBIs have only been developed for a limited set of systems. This situation is discussed in Carter et al. (2019), who reveal an imbalance between a preponderance of ecological integrity measures for aquatic ecosystems and a paucity of similar measures for terrestrial systems. Thus, lack of metric development may be a barrier to the evaluation and use of these measures in many instances, particularly for terrestrial settings.

The literature also provides longstanding evidence that commonly used measures of environmental quality in ecosystems—such as WQIs in water bodies and various measures of air quality and visibility—can be relevant to use and nonuse values. Also relevant are characteristic quantity measures such as instream flow within rivers. Although many measures of this type exist across the literature (too many to catalog here), there is particularly strong and consistent evidence for a few types of quality and quantity measures. These include various types of water quality ladders and metrics grounded in underlying WQIs of the type used by US EPA to characterize water quality change (Walsh and Wheeler 2013), along with measures of haze or visibility in iconic areas such as national parks. An advantage of linking indicators of this type is that they may be easily developed for nationwide application, using data that are already available. There is also strong evidence for various types of quantity indicators such as measures of tree density in forests—although the degree to which these are relevant in different types of ecological landscapes is not clear.

In summary, a review of the literature that seeks to elicit existence values associated with changes to ecological landscapes raises a variety of unresolved questions regarding linking indicator development. To date there have been few efforts to systematically compare the performance of different indicators used to quantify the same underlying changes.[55] Nonetheless, our review provides some guidance for those considering linking indicator options for these systems.

7. Conclusions

Accurate quantification of existence value—an often-substantial component of total economic value—is important to environmental policy and management decisions. Within environmental economics, most attention in this area has been devoted to the economic validity and reliability of stated-preference methods used to estimate these values. Less attention has been given to the arguably more fundamental question of how the commodities that produce these values—whether related to species, landscapes, or ecological systems—are best defined and measured. An ability to define, quantify and measure these commodities is a precursor to valid value estimation and the consideration of existence values within policy and program decisions.

This paper proposes a set of core principles to guide the definition and measurement of the commodities that produce existence values. These principles are drawn largely from a review and synthesis of (1) the stated-preference literature focused on estimating economic values linked to these types of commodities and (2) the natural science literature on criteria for biophysical indicator development. We emphasize the joint importance of measures that are both

[55] Examples of this work in the literature include Bateman et al. (2009), Johnston et al. (2017) and Zhao et al. (2013).

biophysically valid and amenable to lay audience comprehension, and provide illustrative examples of measures that meet those criteria. We also provide examples of measures that do not meet these criteria.

Among the most important conclusions of this review is that there is no compact set of indicators or measures that applies universally. Although we propose general guidelines for existence-value commodity measurement that are intended to apply broadly, the definition of these measures and indicators for particular applications (or case studies) depends on the goal(s) of analysis, the decision(s) to be informed, the intended users of the information, the data that are available, and the feasibility of measurement across different settings, among other features. For example, an otherwise ideal indicator may be of little practical utility if the data required to develop it are unavailable for contexts within which the indicator is to be applied. Further confounding the search for "universal measures" is the often compound, bundled nature of ecological commodities and the spatial definition of species populations, landscapes, and ecosystems. These bundles may be understood and valued differently across contexts, and by different beneficiary groups. For example, the particular bundle of measurable characteristics that best defines the essential nature or existence of one ecological landscape (e.g., the Great Barrier Reef) may differ markedly from the bundle that characterizes other, even broadly similar systems (e.g. cold water corals in Norway; Aanesen et al. 2015).

The case-specific nature of these issues—general guidelines notwithstanding—implies the need for sustained partnerships between social and biophysical scientists to address questions of commodity measurement at a granular level. These engagements should focus not only on the operational definition of the metrics themselves, but also on what each metric is intended to convey and to whom, how it is interpreted by different groups (e.g., beneficiaries and decision-makers), and whether different groups share an understanding of how and why the metric is related to existence values. Engagement of this type is essential to ensure that the commodity measures used to inform valuation and decision-making are valid, relevant, and informative from the perspective of both the natural and social sciences.

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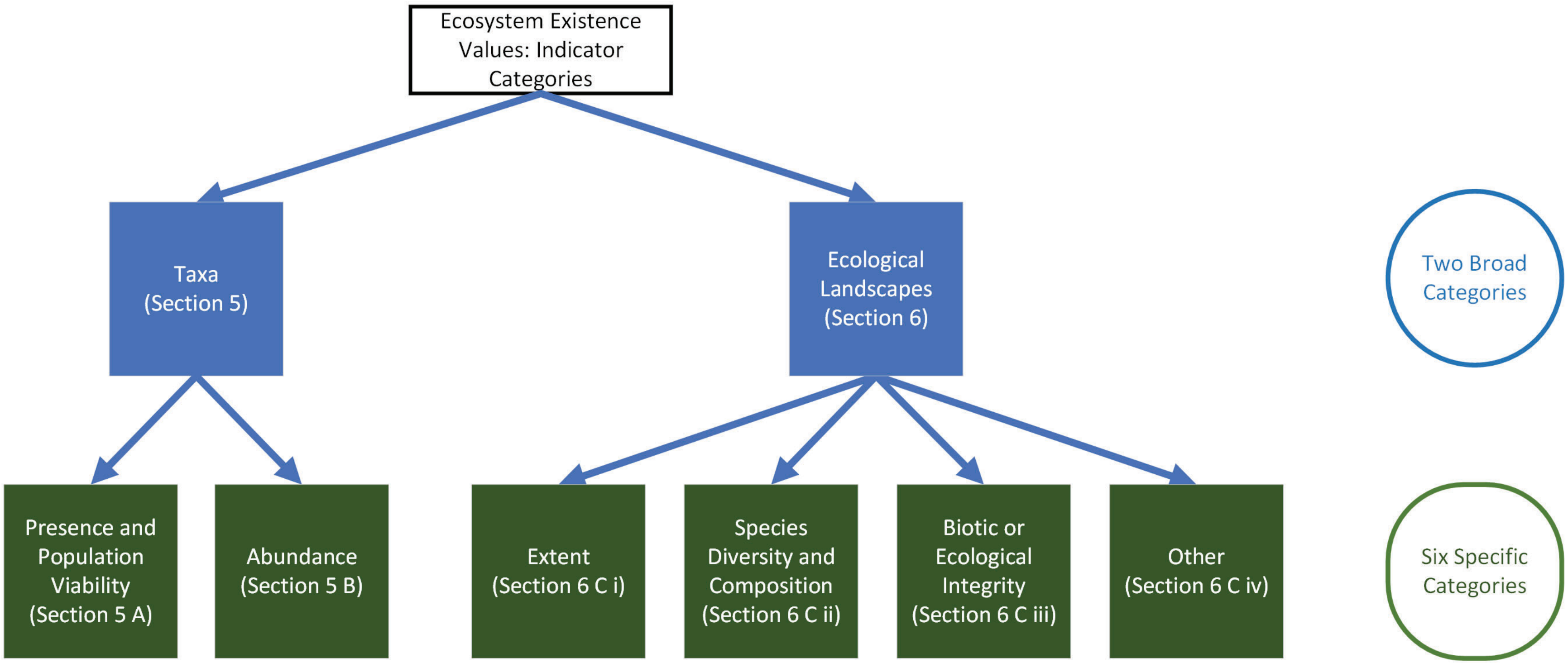

Figure 1. We propose ecological outcomes associated with existence values. Our analysis is organized around two broad categories. Within these, we propose several more specific categories of linking indicators. The text identifies (in Section 3) six criteria for candidate metrics of these linking indicators. Sections 5 and 6 of the analysis identifies candidate metrics that may serve to represent these linking indicators. These candidate metrics are discussed in light of the six criteria applicable to the acceptability of linking indicators.