

**Title** Reference state and benchmark concepts for better biodiversity conservation in contemporary ecosystems

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**Abstract:** Measuring the status and trends of biodiversity is critical for making informed decisions about the conservation, management or restoration of species, habitats and ecosystems. Defining the reference state against which status and change are measured is essential. Typically, reference states describe historical conditions, yet historical conditions are challenging to quantify, may be difficult to falsify, and may no longer be an attainable target in a contemporary ecosystem. We have constructed a conceptual framework to help inform thinking and discussion around the philosophical underpinnings of reference states and guide their application. We characterise currently recognised historical reference states and describe them as Pre-Human, Indigenous Cultural, Pre-Intensification and Hybrid-Historical. We extend the conceptual framework to include contemporary reference states as an alternative theoretical perspective. The contemporary reference state framework is a major conceptual shift that focuses on current ecological patterns and

identifies areas with higher biodiversity values, regardless of the disturbance history. The specific context for which we design the contemporary conceptual frame is underpinned by an overarching goal - to maximise biodiversity conservation and restoration outcomes in existing ecosystems. The contemporary reference state framework can account for the inherent differences in the diversity of biodiversity values (e.g., species richness, habitat complexity) across spatial scales, communities and ecosystems. In contrast to historical reference states, contemporary reference states are measurable and falsifiable. This road map of reference states offers perspective needed to define and assess the status and trends in biodiversity and habitats. Our framework for contemporary reference states provides a tractable way for policy-makers and practitioners to navigate biodiversity assessments to maximise conservation and restoration outcomes in contemporary ecosystems. We illustrate how to define a contemporary reference state using an example from south-eastern Australia.

**Keywords:** Conservation, restoration, reference state, benchmark, vegetation, composition, structure, conceptual framework

## 1. Introduction

A fundamental principle underpinning the conservation, management and restoration of species and habitats is assessing their current state against some former state or baseline. However, choosing 'which state' or 'which baseline' are questions that challenge conservation practitioners. For example, the 2019 global assessment by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) used multiple baselines to assess status and trends. The authors of IPBES (2019) estimate that 47% of ecosystems have declined in extent and condition when compared to 'prehistory baseline'; habitat integrity has reduced by 30% relative to an 'unimpacted baseline'; and that 680 vertebrate species have become extinct 'since the year 1500' (IPBES 2019). Alternatively, the Living Planet Index uses a more recent, 1970 time-stamped baseline, to assess and monitor the decline in the abundance of vertebrate populations and species (Butchart et al. 2010; Tittensor et al. 2010). These global assessments demonstrate some approaches for assessing the status and trends in biodiversity that are urgently needed to galvanise conservation efforts. However, where baselines and reference states are derived from different metrics and span timeframes from many thousands of years to fewer than 50 years, their potential for assessing biodiversity conservation and restoration outcomes in existing ecosystems can be constrained.

Because of the wide-ranging end goals and inconsistent timeframes, the reference state concept has been criticised as being too complicated to quantify and challenging to confirm (Corlett 2016; Hobbs et al. 2014; Hughes et al. 2017; Kopf et

al. 2015; Suding 2011). Where reference states are defined by distant, variable and non-specific historical timeframes, these may represent impractical targets needed to guide contemporary biodiversity conservation (Suding 2011). In addition, current environmental conditions have changed markedly from those that underpin historical reference states, as such, historical reference states may be unsuitable for contemporary and future ecosystems (Hobbs et al. 2014).

Adding to the concerns about defining historical reference states is that the benchmarks (see definitions in Box 1) used to describe historical reference states are often derived from heuristic approaches like best professional judgement or opinion (e.g., Faber-Langendoen et al. 2012). Expert judgement is beneficial when empirical data are scarce, multiple experts can be canvassed, or rapid decision support is needed to help inform policy (e.g., Gibbons et al. 2009; Sinclair et al. 2015). However, expert-defined reference states embody several assumptions regarding disturbance patterns and processes; opinions of experts can be laden with varying historical, social, political, economic and cultural values (Burgman et al. 2011; Kahneman 2011). For example, there is little disagreement that natural and Indigenous-managed fires have influenced vegetation composition and structure but, there are debates and uncertainties about the extent, frequency and severity of fire regimes (Bowman et al. 2011; Denevan 1992; Enright and Thomas 2008; Gammage 2011). These debates are difficult to resolve because historical, archaeological and palaeo-ecological evidence is fragmented and can be difficult to interpret with certainty rendering expert opinion challenging to confirm.

Here we examine the historical reference state concept (see Box 1) to assess biodiversity values in the context of ecosystem conservation and management. We review the applications of historical reference states for assessing site-scaled biodiversity values. We have constructed a conceptual framework to understand the underpinnings of reference states and guide their application (Figure 1). We show how different reference states should be tailored to specific contexts, spatial and temporal scales. We then propose contemporary reference states as an alternative framework to inform better biodiversity conservation and management decisions in contemporary landscapes. The specific context for which we design the contemporary conceptual frame is underpinned by an overarching goal—to maximise biodiversity conservation and restoration outcomes in existing ecosystems. Finally, we illustrate an operational approach to defining site-scaled biodiversity values for contemporary reference states and outline its benefits.

**BOX 1: DEFINITIONS OF TERMINOLOGY USED IN THIS SYNTHESIS**

**Conceptual frame** is the *theoretical perspective* from which reference states are identified and defined. The choice of conceptual frame and the underlying assumptions are context dependent and need to be explicitly defined when assessing species, habitats or biodiversity. At the broadest level in our conceptual framework we define a dichotomy in the context and purpose for reference states, and we split them into either *historical* or *contemporary* (see Figure 1).

**Reference states** also referred to as reference condition, is the *ecological context* for how we assess the current state of species, populations, habitats, ecosystems or biodiversity. Reference states can be specified using site-scaled or landscape-scaled proxies, surrogates or indicators

**Baselines** define the *temporal dimension* across which we assess change. Timeframes for baseline reporting need to be clearly defined although the temporal specificity of the baseline can differ for different end goals.

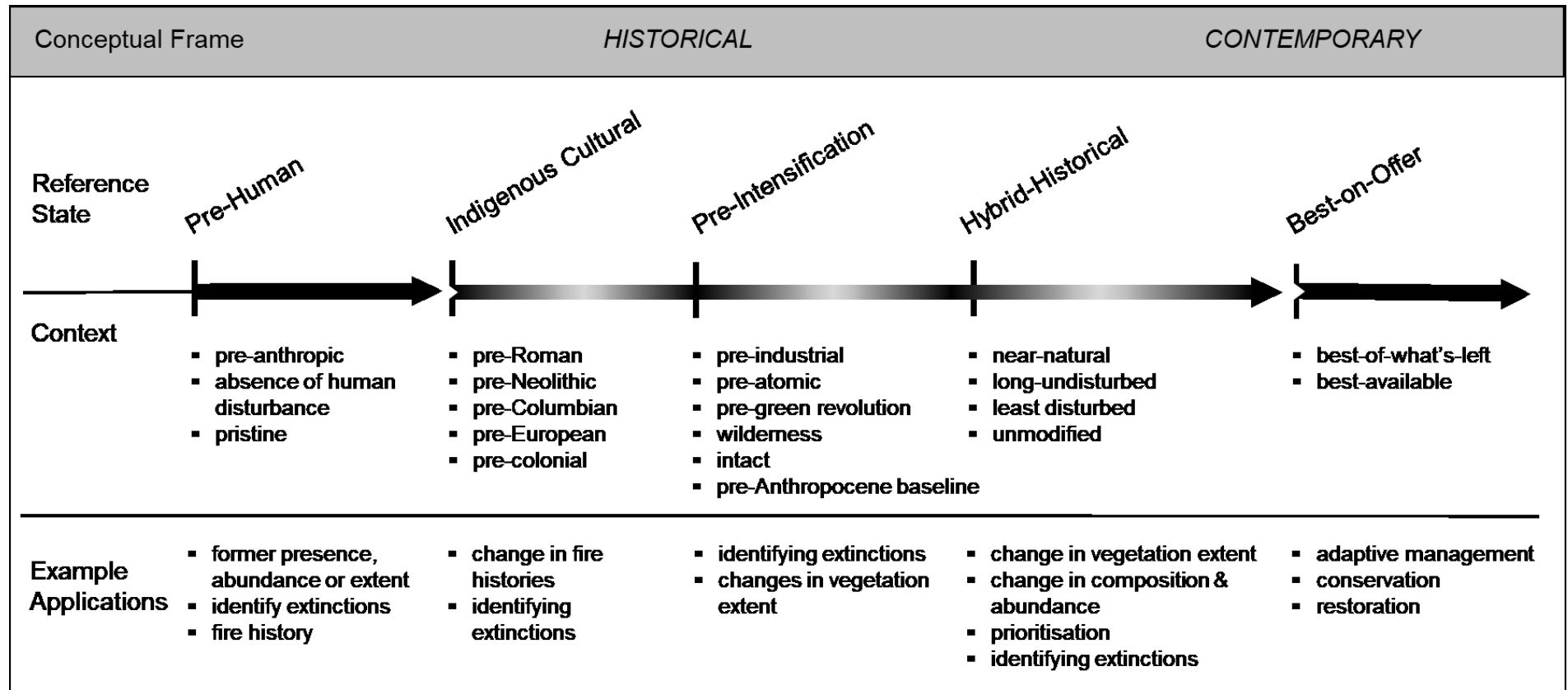
**Benchmarks** are the *numerical context* that defines the reference state. These may be empirically- or expert-derived from abundance, richness or diversity of genes, populations, species, communities or their habitat, or from composite measures of biodiversity composition, structure or function (Noss 1990).

**2. Reviewing the historical reference state conceptual frame**

Traditionally, thresholds of transformative change define historical reference states; and most often anthropogenic disturbances are the drivers of change. Here we extend the concepts proposed by Stoddard et al. (2006) and identify four distinct historical reference states as Pre-Human, Indigenous Cultural, Pre-Intensification

and Hybrid-Historical (Figure 1). As these reference states are conceptual, they do not necessarily have specific date stamps, nor do they have rigid or fixed temporal windows. They represent a continuum of human disturbance, and thus different types of reference states are used for conservation, management and restoration. A principal tenet of our conceptual framework is that we recognise ecosystems have transitioned through these disturbance intervals, and thus ecosystems carry a cumulative and historical legacy of change.

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3 Figure 1: A conceptual framework for synthesising the historical and contemporary reference states and their context and  
 4 applications within the context of informing biodiversity conservation and restoration outcomes in existing ecosystems.



### *Pre-Human reference state*

Pre-Human reference states represent the temporal baseline before human presence. Often Pre-Human reference states are regarded as pristine and are synonymous with concepts such as or pre-anthropogenic, historical or natural range of variability (Landres et al. 1999; Stoddard et al. 2006) and potential natural vegetation (Loidi and Fernández-González 2012). For ecosystem conservation, management and restoration the Pre-Human baseline can be used to evaluate changes in the extent of habitat (Zerbe 1998) and the composition and abundance of species (Lotze and Worm 2009; Zu Ermgassen et al. 2012), disturbance regimes and environmental conditions. Most often, Pre-Human baselines are derived from palaeo-ecological evidence. For example, sedimentary charcoal data infer fire variability (Floyd and Willis 2008) and stable isotope and geochemical studies have helped reconstruct patterns in climate variability (Birks 2012). Palaeo-ecological information has provided a better understanding of how Pre-Human reference states have been transformed as a result of overharvesting, overhunting, habitat transformation and mass extinctions (Burney and Flannery 2005). Floyd and Willis (2008) suggest that Pre-Human reference states represent long-established patterns where ecosystems were in equilibrium with past environments. However, in most instances, inference based on Pre-Human reference states is too sparse to be used for broader landscape-scaled reconstructions of historical ecosystems (Seddon et al. 2014).

### *Indigenous Cultural reference state*

The Indigenous Cultural reference state represents the time-period between the arrival of First Nations and Indigenous people, up to Roman, Columbian or European colonisation. We classify pre-colonial, pre-Columbian, pre-European and pre-1610

Orbis spike as Indigenous Cultural reference states. The conceptual frame for Indigenous Cultural reference state aims to recognise the transition in land management practices and the consequent and intensified impact on the natural environment. Bliege Bird and Nimmo (2018) purport that Indigenous place-based societies co-evolved with ecosystems across vast portions of the globe. Indigenous cultural management practices such as harvesting or burning (Anderson 2005; Kimmerer 2002), trapping and hunting (Rose et al. 2016) or landscape scaled fire management (Bowman et al. 2011; Denevan 1992; Gammage 2011) aim to incorporate ecosystem processes such as herbivory, seed dispersal, soil turnover, predation and burning to maintain conservation, management and restoration (Bjorkman and Vellend 2010). By restoring sophisticated and strategic traditional land use practices, Indigenous Cultural reference states seek conservation strategies that manage fire, invasive species and biodiversity loss (e.g., Ban et al. 2020; Ens et al. 2015; Molnár et al. 2020).

These reference states, which include the pre-Columbian reference applied in the Americas and European countries, or pre-European applied in Oceania and Canada, mark disturbances related to introducing non-native or alien species, especially plant and animal species used as food sources (Kühn et al. 2004; Pyšek et al. 2004) and altered landscape configuration and composition as a direct result of acclimation and agricultural practices. These reference states have been used to report on the extent of vegetation change in Australia (e.g., Bradshaw 2012; Gibbons et al. 2009; Mendel 2002), Africa (e.g., Aleman et al. 2018; Cowling et al. 2003) and North America (e.g., Keddy and Drummond 1996; Motzkin and Foster 2002). There are many examples where former climates, landscapes and vegetation have been

extrapolated from palynology, peat, sediment, or ice cores. Some clear evidence of pre-Columbian and pre-European reference states can be derived from historical survey records, tree stumps or pattern in vegetation clearing for agriculture (e.g., Aleman et al. 2018; Bjorkman and Vellend 2010; Fensham and Fairfax 1997; Jackson 2013; Lunt 2002; Seabrook et al. 2006; Silcock et al. 2013; White and Mladenoff 1994). The Indigenous Cultural reference states are tractable because there are often clear connections between altered disturbance patterns induced by non-Indigenous colonisation and the decline in 'naturalness' and an increase in extinction rates and invasion by non-native species.

#### *Pre-Intensification reference states*

The Pre-Intensification reference state represents critical temporal break-points prior to amplified rates of change attributed to human disturbance (Heller and Hobbs 2014; Lewis and Maslin 2015). For example, industrialisation and extraction of fossil fuels have triggered global transformative changes (Steffen et al. 2011). However this reference state is not confined to impacts resulting from the Industrial Revolution. Evidence of the long history of human use of metals and subsequent metal pollution such as copper smelting used during Roman Empire (circa 2 000 yr before present) can be detected in ice cores (Lewis and Maslin 2015). Although intensification initially commenced in localised areas, impacts of human modification and disturbance can be detected using local and global environmental indicators.

In contrast to the Pre-Human and Indigenous Cultural reference states, baselines for Pre-Intensification reference states can be clearly defined using environmental markers at global, regional and local scales. For example, spikes in radionucleotides

and atmospheric  $^{14}\text{C}$  determine the onset of global intensification (Lewis and Maslin 2015; Turney et al. 2018). Measuring environmental markers, such as nitrogen and phosphorous from fertilisers (Lu and Tian 2014) or heavy metals (Shotbolt et al. 2007), offers a tractable approach to identifying Pre-Intensification reference states. For example, the European Water Framework Directive criteria define “concentrations of specific synthetic pollutants should be close to zero” as a concrete descriptor of Pre-Intensification reference states, and these markers are easier to measure compared with other criteria, such as biotic integrity (Hering et al. 2010).

### *Hybrid-Historical reference states*

The Hybrid-Historical reference state represents a reconstruction of past conditions based on current patterns. This reference state moves towards bridging historical and contemporary theoretical perspectives (Figure 1) and perceives historical states through a contemporary lens. Hybrid-Historical is synonymous with ‘near-natural’, ‘long-undisturbed’, ‘unmodified’, ‘relatively undisturbed’, or ‘least-disturbed’.

Underpinning the Hybrid-Historical reference state is an assumption that least disturbed ecosystems retain a majority of native biota (Landres et al. 1999); are most resilient to disturbance (Holling 1973; Naeem and Li 1997; Yachi and Loreau 1999) and thus support better conservation outcomes. Unlike Pre-Human, Indigenous Cultural and Pre-Industrial, the Hybrid-Historical reference state is not defined by time stamps and may combine evidence from multiple reference states shown in Figure 1. For example, Hybrid-Historical reference states can be described from on-ground evidence in conjunction with evidence of post-intensification anthropogenic disturbance, such as vegetation clearing, timber harvesting, the extent of change in the composition of vegetation, cultivation, fertiliser application, livestock grazing or

historical aerial photography. Often this reference state is identified by combining both current and reconstructed historical patterns of disturbance to identify least-disturbed and long-undisturbed reference states (e.g., Gibbons et al. 2008; Hessburg et al. 1999; Seddon et al. 2011). An alternative approach to measuring on-ground evidence of disturbance history has been the use of land tenure (such as protected areas, nature reserves and national parks) as a proxy (e.g., Scholes and Biggs 2005; Sinclair et al. 2002). Landscape condition assessments have used national conservation reserves (e.g., Harwood et al. 2016) or identified areas based on their distance from human populations or infrastructure (e.g., Allan et al. 2019; Watson et al. 2018) as the benchmark for identifying least disturbed. However, protected areas are not always minimally disturbed; are often not representative of a majority of ecosystems (Joppa and Pfaff 2009). Some areas with the greatest biodiversity value are in unprotected tenures (Archibald et al. 2020; Clements et al. 2019; De Vos and Cumming 2019) and face some of the greatest threats (Myers et al. 2000).

Hybrid-Historical reference states can be differentiated from Indigenous Cultural reference states in that they do not explicitly account for prior Indigenous ecological management practices and how this may have influenced biotic relationships with climate and landscape. The Hybrid-Historical reference state is appealing because it can overcome some of the uncertain or unquantifiable characteristics of historical ecosystems (Balaguer et al. 2014).

### **3. Limitations of historical reference states**

Historical perspectives of ecosystems gleaned from multiple lines of evidence have been essential for developing hypotheses for how ecosystems evolve and function (Swetnam et al. 1999). Historical drivers of change have been vital in navigating how ecological processes determine contemporary patterns of biodiversity and predicting changes in ecosystems. However, historical reference states do not explicitly target the goal of maximising biodiversity conservation outcomes in contemporary ecosystems; at best they hypothesise that attaining a historical reference state will have a co-benefit of maximising biodiversity. Within the specific context of setting targets for maximising biodiversity through ecosystem management, conservation and restoration actions, we argue that there are three key limitations to the use of historical reference states in contemporary ecosystem management. Firstly, historical reference states are often unmeasurable. Secondly, they are almost always unfalsifiable. Finally, historical reference states may be unattainable in contemporary landscapes. While this final point is not important in the context of describing change over time, it is a significant limitation in the context of setting targets for maximising biodiversity outcomes through ecosystem management, conservation and restoration actions.

#### *Historical reference states are often unmeasurable*

One of the key criticisms of historical reference states is that they are difficult to quantify. Prior ecological patterns and their associated climate, disturbance regimes and biological interactions may not be represented in present-day analogues.

Therefore, historical reference states must be indirectly inferred, extrapolated or reconstructed, often from fragmented evidence. Long-term, detailed, continuous data are lacking for most species, communities, ecosystems and few population trends have been studied for more than 100 years (Bonebrake et al. 2010; Mihoub et al.

2017; Vihervaara et al. 2013). Detailed, site-specific biological records rarely pre-date human disturbances (Hobbs et al. 2010, Balaguer et al. 2014) and where long-term records have been collected, rarely do they adequately describe detailed community structure and species composition. As a result, reconstructions of species' assemblages or ecosystems may be too imprecise, incomplete or inadequate to inform ecosystem-level decisions (Stoddard et al. 2006) and are likely to be biased given the non-random nature of human disturbance.

Because historical reference states and their benchmarks are often pieced together from sparse information, they can be reliant on expert judgement. These opinions can be uncertain, value-laden, subjective and prone to counter-perspective and discrepancy (Burgman et al. 2011). The bias, validity and reliability of expert judgement are especially problematic when it is difficult to assess the transparency and repeatability of the methods (Drescher and Edwards 2019). Consequently, reproducing benchmarks generated by expert opinion may lead to discrepant results, wide variations in estimates or contested debate.

In some contexts, time-stamps can also be challenging to quantify or confirm. Pre-Human and the subsequent Indigenous Cultural reference states are difficult to pinpoint with certainty because there have been multiple waves of both Indigenous and non-Indigenous colonisation (Dubois et al. 2018) and these patterns have not been uniform across whole landmasses. Locally and regionally, reference states would have varied time-stamps owing to the radiation of human migrations, and therefore a specific time-stamp is ill-defined and difficult to discern. As an example, Dutch and Spanish and Macassan people arrived on the Australian continent as

early as 1606, well before 1788 British colonisation. The subsequent patterns of British colonisation differed over all parts of the continent. In contrast, Pre-Intensification reference states can often be time-stamped consistently and accurately using environmental markers and can be correlated with ecological or biotic patterns relevant to biodiversity conservation and management (Zhang et al. 2018).

The Hybrid-Historical reference state neither depends on explicit time-stamps nor estimates of historical structure and composition of biota. However, it does rely on consistently defining ecosystems that are least disturbed and quantifying disturbance history, which can be challenging and a key source of uncertainty (Nagel et al. 2007). Hybrid-Historical reference states also rely on space-for-time as a substitute for temporal data (Cava et al. 2018; Lotze and Worm 2009; Symstad and Jonas 2014). Space-for-time experimental designs are common but criticised because they assume ecological observations are at equilibrium (Damgaard 2019). However, this assumption relies on knowledge of temporal dynamics which are difficult to estimate in the absence of long-term data (De Palma et al. 2018). Space-for-time substitutions are difficult because they require large unaltered areas or remote intact regions that represent the range of conditions in the contemporary ecosystem. However, large, intact, remote and protected ecosystems are often temporally and spatially biased (Damgaard 2019; Joppa and Pfaff 2009; Negret et al. 2020) and do not account for non-random human disturbance.

*Historical reference states are often unfalsifiable*



Because historical reference states are often unmeasurable, they generally cannot be tested, evaluated and improved by falsifiable enquiry (McCarthy et al. 2004). This is a critical shortfall of most historical reference states because the foundation of transparent and rigorous science is repeatable methods and measurements (Wolman 2006). In the absence of a systematic and comprehensive set of empirical samples to describe historical reference states, it is problematic to prove any meaningful relationships between biodiversity values and reference states. We highlight that Pre-Human reference states are the least likely to be falsified and the Hybrid-Historical reference state approaches the premise of falsifiability. In Hybrid-Historical reference states, ecosystems can be identified as 'undisturbed' and hypotheses can be tested to ascertain if higher biodiversity values are observed within these Hybrid-Historical reference states.

*Historical reference states may be inappropriate because they are unattainable*

Most of the Earth's ecosystems have been adversely affected by anthropogenic pressures (Butchart et al. 2010). When historical reference states are perceived through a lens of either ecological or social values or both, then returning to the former historical state may sometimes be unattainable. For example, Zweig and Kitchens (2010) argue that aiming to restore the Florida Everglades to a pre-1880 historical state, prior to extensive changes in the hydrology, vegetation and soils, would adversely impact on current livelihoods, food security, economies and societal conditions. In instances such as this, proliferating population and intensifying urban and agricultural development cannot be reconciled with the restoration of historical states. The ecological-societal nexus conflicts with the potential to 'return' to a historical reference state.

Current biological and abiotic pressures can also hinder passive recovery or make efforts to pursue active restoration or re-introductions untenable. Where the global biotic and abiotic conditions (such as climate, changes in flood regimes, changed fire intensity and frequency, warming ocean temperature, rising sea level, increased atmospheric CO<sub>2</sub> or overexploitation) cannot be reinstated, or are poorly known, historical reference states may be unattainable and unrepresentative of contemporary habitats or biodiversity (Corlett 2016; Rohwer and Marris 2016). In some instances local abiotic conditions have moved beyond remediation (e.g., altered soil properties, nutrient, pH) or biotic assemblages (habitat alteration and loss, increase in invasive species, loss of ecosystem-engineering species or extinct mega-fauna) cannot be reinstated. For example, as a result of rising sea temperatures, Hughes et al. (2017) consider the recent historical conceptual frame as a non-viable option for managing contemporary coral reefs. Similarly, Bond and Midgley (2012) postulate that historical fire regimes may be inadequate for maintaining tree-grass dynamics in humid savannah ecosystems as a result of the near doubling of atmospheric CO<sub>2</sub>.

Hobbs et al. (2014) argue that many ecosystems have so radically changed from any recognisable historical reference state that conservation outcomes in contemporary ecosystems need an alternative approach for setting conservation management and restoration goals. By classifying ecosystems based on irreversibility, Hobbs et al. (2014) provide a framework for evaluating where and if historical reference states are attainable.

Where societal, abiotic and biological constraints can be reversed, ecosystems could be managed as hybrid ecosystems based on historical reference states. Whereas ecosystems that arise as a result of irreversible barriers, may need to be managed as novel ecosystems with no clear historical analogue. This approach is a useful guide to construct a decision framework as it outlines how management goals can be evaluated. Where the conservation, management and restoration goal is to maximise biodiversity conservation and restoration outcomes for contemporary ecosystems, consideration and decoupling of the societal and biological constraints will help determine feasible and appropriate applications and inform how, where and if historical reference states are attainable.

#### **4. An alternative theoretical perspective – contemporary reference states**

To overcome the complications and limitations associated with defining and measuring historical reference states, and with maximising biodiversity conservation and restoration outcomes for contemporary ecosystems, we promote an alternative theoretical perspective based on the concept of ‘contemporary reference state’ (Figure 1). A transition away from historical disturbance-driven reference states to contemporary biodiversity value-based reference states signals an opportunity to assess diversity-driven characteristics of contemporary ecosystems and overcome the complications and limitations of the disturbance-driven approach. Kopf et al. (2015) also recognise the benefits of contemporary reference states and propose Anthropocene baselines as an alternative framework for contemporary ecosystems. However, unlike Kopf et al. (2015), we propose a shift from a disturbance-driven philosophy to a diversity-driven philosophy.

Our conceptual framework for contemporary reference states extends the 'Best-of-What's-Left' concept (Stoddard et al. 2006) in that it doesn't contemplate historical reference states, but orientates to known, extant states that are identified by high levels of current biodiversity values within and among contemporary ecosystems. The concept centres on a 'more diversity is better' philosophy and builds on the 'Best-on-Offer' reference state proposed by Eyre et al. (2006) whereby biodiversity values that exist under contemporary conditions can be clearly defined and represent an approach to quantifying the status of biodiversity.

Given no community consists of the same species in equal abundance, we argue the contemporary reference state concept should be based on a standardised assessment of native species diversity and measured relative to the native species diversity within the same ecosystem with consistent climatic, edaphic, topographic and biogeographic characteristics (Sinclair et al. 2002; Symstad and Jonas 2014). By assessing native species diversity using standardised approaches and relative to a comparable community or ecosystem, the contemporary reference state approach can account for the inherent turnover in biodiversity that occurs between spatial scales, communities and ecosystems. That is, contemporary reference states and the benchmarks that define them are specific to ecological communities assessed at consistent spatial scales. Box 2 illustrates one recent attempt to identify and operationalise the contemporary Best-on-Offer reference state. In this operational example, sites with higher numbers of native plant species and greater structural complexity (relative to other sites of the same vegetation type) were used to identify the Best-on-Offer reference state as an approach to improving biodiversity conservation outcomes within south-eastern Australia (Yen et al. 2019).

The ecological justification for describing a Best-on-Offer reference state can be supported by theoretical underpinnings that hypothesise that a site with higher levels of alpha-diversity has increased stability over time (Hector et al. 2010; Loreau et al. 2001; McCann 2000; Tilman et al. 2001; Tilman et al. 2006); stability over space (Hector et al. 2010; McCann 2000); and adaptive capacity (Baho et al. 2017). Sites with greater alpha-diversity have greater ecological integrity (Brooks et al. 1998; Oliver et al. 2014); higher productivity (Tilman et al. 2012); and provide a greater diversity of ecosystem services (Hooper et al. 2005) and functions (Schwartz et al. 2000). The 'more diversity is better' paradigm moves towards an insurance policy approach and assumes where ecosystems are replete they are more resilient (Holling 1973; Naeem and Li 1997; Yachi and Loreau 1999). However, by defining the contemporary reference state at the biome, ecoregion or ecosystem level, the concept also accounts for beta- and gamma-diversity (Whittaker 1967).

## **5. Benefits of diversity-driven contemporary reference states**

A diversity-driven contemporary reference state approach has a number of strengths. It does not need to make assumptions about disturbance history, natural or historical range of variability, nor the interactions between human-induced and natural disturbances. Importantly, diversity measures are comparative within the same ecological units which can account for describing reference states in different conditions, including naturally depauperate ecosystems. Contemporary reference states can be specifically defined across time and space (Balaguer et al. 2014).

In addition, the contemporary reference state concept can potentially overcome some of the shortfalls we have outlined with the historical reference state concept because the diversity-driven approach allows us to quantify biodiversity values using empirical data collected from within existing ecosystems. It does not require expert opinion to estimate what is 'good' quality or 'higher' integrity or 'maximum' biodiversity value. It circumvents the problems with using expert opinion, in which evaluation of benchmarks is based on subjective counter-opinion (Sinclair et al. 2015). The empirical description of reference states enables a more transparent, rigorous, repeatable and falsifiable approach to assessing conservation and restoration targets (Box 1) and moves us away from a value-laden approach to evaluating ecosystems (Wolman 2006). This is important when comparing different biodiversity outcomes that may be competing or contentious.

Several additional benefits of shifting to a Best-on-Offer reference state include the ability to locate reference sites spatially and temporally. Reference sites can be identified from underlying data and can conceivably become the 'holotypes' that demonstrate on-ground characteristics of the Best-on-Offer reference state. Holotype sites can be measured and monitored to gain a better understanding of how biodiversity composition, structure and function at reference sites change over time (Hiers et al. 2012). This could include year-to-year variation as well as longer-term trends that might be associated with large scale shifts in climate, biotic interactions or disturbance regimes. Furthermore, benchmarks can be refined by ongoing assessment and monitoring of holotype sites which has benefits of delivering the potential for dynamic benchmarks that represent seasonal and climatic changes (Yen et al. 2019). The timeframe of the reference state baseline is clearly articulated

because this approach is based on the underlying data. Therefore benchmarks to define the contemporary reference states can develop and improve as new data become available. Moreover, the contemporary reference state concept is scalable, and where data are available, this approach can be applied at local, regional and global scales providing that reference states are defined relative to individual ecological units.

## **BOX 2 - MEASURING DIVERSITY-DRIVEN CONTEMPORARY REFERENCE STATES**

### **An operational approach for defining contemporary Best-on-Offer reference states to support the Biodiversity Conservation Act in New South Wales, Australia**

Foliage cover and species richness of native plants provides information on vegetation structure and composition and are frequently used as surrogates for biodiversity (Westgate et al. 2018). With the ever-expanding repositories of floristic data that contain inventories of plant species, there are opportunities to develop empirical benchmarks using existing vegetation data. Here, we demonstrate how floristic data were used to define the Best-on-Offer benchmarks to support and implement the Biodiversity Assessment Method under the Biodiversity Conservation Act 2016 (Yen et al. 2019). Under the New South Wales Biodiversity Conservation Act 2016, Vegetation Integrity was identified as a key biodiversity value and defined according to benchmarks for foliage cover and richness of native plant growth forms along with other habitat attributes.

## Screen and Stratify

From the BioNet Atlas ([www.bionet.nsw.gov.au](http://www.bionet.nsw.gov.au)) we extracted approximately 70 000 archived floristic plots. We screened these data and selected only plots that were comprehensively and systematically surveyed within a fixed 0.04 ha area (approximately 36 000 plots). We stratified the study area by geomorphological units (Figure 2) and vegetation class to create regional vegetation classes (RVCs).

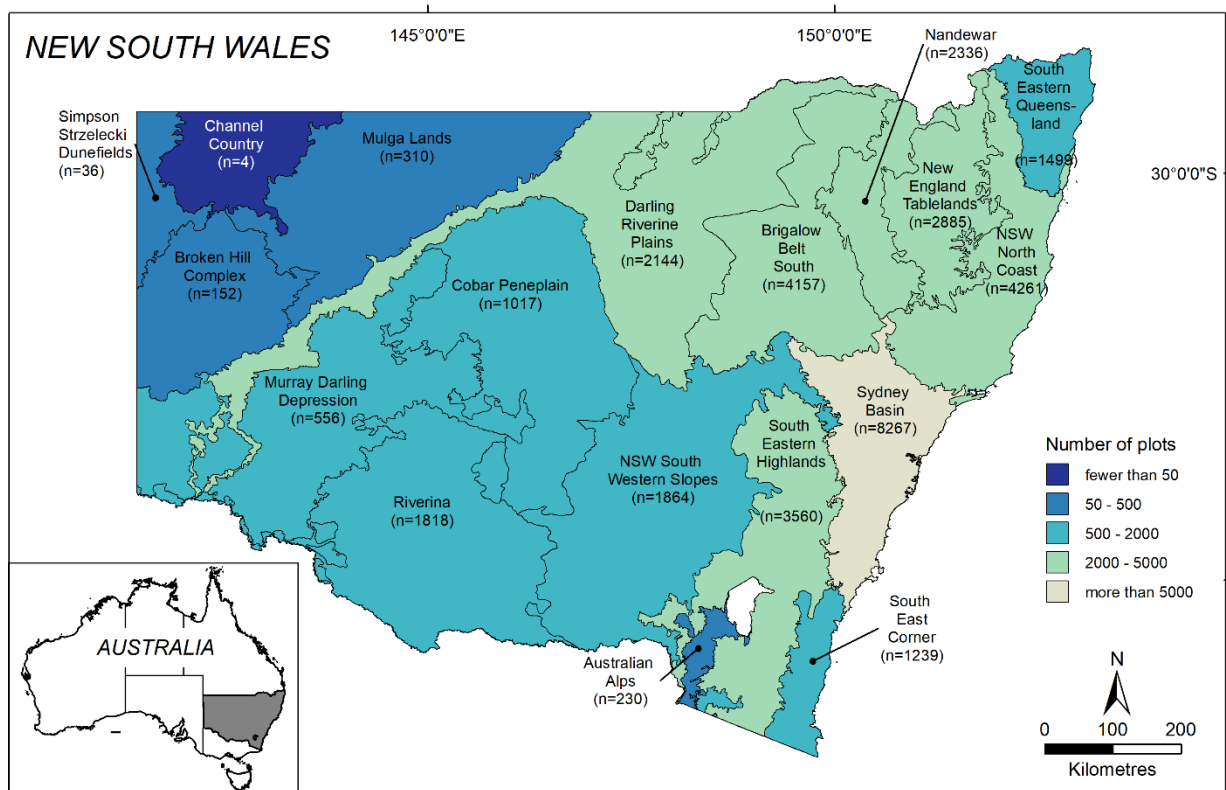


Figure 2: Eighteen geomorphological units used in combination with vegetation class to stratify the study area. Numbers shown in parentheses are the number of floristic plots after data screening.



### *Synthesise and operationalise*

For each of the approximately 36 000 plots we allocated all native vascular plant species to one of six growth forms trees, shrubs, grasses, forbs, ferns and others (Oliver et al. 2019) and calculated the structural (summed foliage cover of species within each growth form) and compositional (species richness within each growth form) attributes for each growth form for each plot. We used modelled upper quantiles of the data distributions to estimate contemporary, Best-on-Offer benchmarks for growth form cover and richness (Yen et al. 2019). Importantly, benchmarks are specific to the structure and composition of growth forms and vary across the landscape according to the geomorphology and vegetation community. Further information is provided at <https://www.environment.nsw.gov.au/topics/animals-and-plants/native-vegetation/vegetation-condition-benchmarks>

This case study demonstrates how plot level data can be used to describe a contemporary Best-on-Offer reference state to maximise biodiversity conservation and restoration outcomes in existing ecosystems and could be extended, using alternative data sources to account for beta- or gamma-diversity between spatial scales, communities and ecosystems.

## **6. Limitations of the diversity-driven contemporary reference state approach**

One of the perceived shortfalls of a contemporary reference state is accepting that the best representation of some ecosystems are altered or degraded, and not representative of an 'ideal' state, and therefore a contemporary conceptual framework lowers or compromises outcomes for biodiversity (Soga and Gaston 2018). We acknowledge these concerns (sensu shifting baseline syndrome Pauly 1995) and argue that the application of contemporary benchmarks requires their explicit consideration. As shown in Box 2, an approach to confirming change in baselines

would be to initiate a network of contemporary reference sites to measure trends in diversity. Describing a reference state using a data-driven approach offers an opportunity to debate or confirm the idea that contemporary reference states are quantifiable. Villnäs and Norkko (2011) argue that shifting baselines as an inevitable consequence for all ecosystems across time and space, but suggest that diversity-driven baselines may offer a transparent solution.

### *Describing patterns, not processes*

One of the most challenging aspects of defining a contemporary reference state is to understand the complex nature of drivers of change. Critically, a contemporary benchmark paradigm does not suggest ecologists should ignore prior patterns and processes. Central to this endeavour is an improved understanding of long term dynamics of ecosystems shaped by slow (such as climate change) and fast (such as rapid socioeconomic development) drivers over various timescales. Without an understanding of both past and present ecological processes, contemporary reference states merely describe observed patterns. That is, we acknowledge that past processes play an essential role in driving contemporary patterns of diversity. Therefore the local conditions in which contemporary reference sites occur, and past disturbances and management to which these sites have been exposed, provide insight into the conditions under which the contemporary reference state may be attained. As outlined above, the contemporary reference state might still be unachievable at some locations because underlying processes cannot be reinstated, or are outside of human influence.

## **5. Conclusions**

Here we synthesise a conceptual framework and provide a 'road map of reference states' to clarify an approach to define and assess the current status and better manage the expected trends in the decline of biodiversity and degradation of habitats. When choosing 'which state' or 'which baseline' we show that different reference states are used in different contexts. Distant historical reference states mark a baseline to assess former presence and abundance of species, identify extinctions, reconstruct former habitats or reinstate past disturbance regimes. Where reference states use more current information and are centred on less distant timeframes, more refined knowledge on structure, composition and function can be generalised. Yet, even when more sophisticated information is available, historical reference states may not best address the need to maximise biodiversity outcomes or inform restoration goals in contemporary ecosystems. The contemporary reference state framework is a major conceptual shift that focuses on current ecological patterns and prioritises conservation of areas with higher biodiversity values. In contrast to historical reference states, contemporary reference states are measurable and falsifiable and can be tested to assess if they do represent high values of biodiversity.

Human-dominated ecosystems have undergone rapid and extensive transformations in the past 50 years and are forecast to accelerate over the next 50 years (Millennium Ecosystem Assessment 2005). We foresee contemporary reference states as an operational tool for policy-makers and practitioners needing to assess biodiversity values consistent with maximising biodiversity conservation and restoration outcomes in contemporary ecosystems.

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## References

Aleman, J.C., Jarzyna, M.A., Staver, A.C., 2018. Forest extent and deforestation in tropical Africa since 1900. *Nature Ecology & Evolution* 2, 26-33.

Allan, J.R., Watson, J.E.M., Di Marco, M., O' Bryan, C.J., Possingham, H.P., Atkinson, S.C., Venter, O., 2019. Hotspots of human impact on threatened terrestrial vertebrates. *PLOS Biology* 17, e3000158.

Anderson, K., 2005. *Tending the Wild: Native American knowledge and the management of California's natural resources*. University of California Press, Berkeley

Archibald, C.L., Barnes, M.D., Tulloch, A.I.T., Fitzsimons, J.A., Morrison, T.H., Mills, M., Rhodes, J.R., 2020. Differences among protected area governance types matter for conserving vegetation communities at-risk of loss and fragmentation. *Biological Conservation* 247, 108533.

Baho, D.L., Allen, C.R., Garmestani, A., Fried-Petersen, H., Renes, S.E., Gunderson, L., Angeler, D.G., 2017. A quantitative framework for assessing ecological resilience. *Ecology and Society* 22, 1-26.

Balaguer, L., Escudero, A., Martín-Duque, J.F., Mola, I., Aronson, J., 2014. The historical reference in restoration ecology: re-defining a cornerstone concept. *Biological Conservation* 176, 12-20.

Ban, N.C., Wilson, E., Neasloss, D., 2020. Historical and contemporary indigenous marine conservation strategies in the North Pacific. *Conservation Biology* 34, 5-14.

Birks, H.J.B., 2012. Ecological palaeoecology and conservation biology: controversies, challenges, and compromises. *International Journal of Biodiversity Science, Ecosystem Services & Management* 8, 292-304.

Bjorkman, A., Vellend, M., 2010. Defining historical baselines for conservation: ecological changes since European settlement on Vancouver Island, Canada. *Conservation Biology* 24, 1559-1568.

Bliege Bird, R., Nimmo, D., 2018. Restore the lost ecological functions of people. *Nature Ecology & Evolution* 2, 1050-1052.

Bond, W.J., Midgley, G.F., 2012. Carbon dioxide and the uneasy interactions of trees and savannah grasses. *Philosophical Transactions of the Royal Society B: Biological Sciences* 367, 601-612.

Bonebrake, T.C., Christensen, J., Boggs, C.L., Ehrlich, P.R., 2010. Population decline assessment, historical baselines, and conservation. *Conservation Letters* 3, 371-378.

Bowman, D.M.J.S., Balch, J., Artaxo, P., Bond, W.J., Cochrane, M.A., D'Antonio, C.M., . . . Swetnam, T.W., 2011. The human dimension of fire regimes on Earth. *Journal of Biogeography* 38, 2223-2236.

Bradshaw, C.J.A., 2012. Little left to lose: deforestation and forest degradation in Australia since European colonization. *Journal of Plant Ecology* 5, 109-120.

Brooks, R.P., O'Connell, T.J., Wardrop, D.H., Jackson, L.E., 1998. Towards a Regional Index of Biological Integrity: The Example of Forested Riparian Ecosystems, In *Monitoring Ecological Condition at Regional Scales: Proceedings of the Third Symposium on the Environmental Monitoring and Assessment Program (EMAP)* Albany, NY, U.S.A., 8-11 April, 1997. eds S. Sandhu, L. Jackson, K. Austin, J. Hyland, B. Melzian, K. Summers, pp. 131-143. Springer Netherlands, Dordrecht.

Burgman, M.A., McBride, M., Ashton, R., Speirs-Bridge, A., Flander, L., Wintle, B., . . . Twardy, C., 2011. Expert status and performance. *PLoS ONE* 6, e22998.

Burney, D.A., Flannery, T.F., 2005. Fifty millennia of catastrophic extinctions after human contact. *Trends in Ecology & Evolution* 20, 395-401.

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., . . . Watson, R., 2010. Global biodiversity: indicators of recent declines. *Science* 328, 1164-1168.

Cava, M.G.B., Pilon, N.A.L., Ribeiro, M.C., Durigan, G., 2018. Abandoned pastures cannot spontaneously recover the attributes of old-growth savannas. *Journal of Applied Ecology* 55, 1164-1172.

Clements, H.S., Kerley, G.I.H., Cumming, G.S., De Vos, A., Cook, C.N., 2019. Privately protected areas provide key opportunities for the regional persistence of large- and medium-sized mammals. *Journal of Applied Ecology* 56, 537-546.

Corlett, R.T., 2016. Restoration, reintroduction, and rewilding in a changing world. *Trends in Ecology & Evolution* 31, 453-462.

Cowling, R.M., Pressey, R.L., Rouget, M., Lombard, A.T., 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. *Biological Conservation* 112, 191-216.

Damgaard, C., 2019. A critique of the space-for-time substitution practice in community ecology. *Trends in Ecology & Evolution* 34, 416-421.

De Palma, A., Sanchez-Ortiz, K., Martin, P.A., Chadwick, A., Gilbert, G., Bates, A.E., . . . Purvis, A., 2018. Chapter Four - Challenges With Inferring How Land-Use Affects Terrestrial Biodiversity: Study Design, Time, Space and Synthesis, In *Advances in Ecological Research*. eds D.A. Bohan, A.J. Dumbrell, G. Woodward, M. Jackson, pp. 163-199. Academic Press.

De Vos, A., Cumming, G.S., 2019. The contribution of land tenure diversity to the spatial resilience of protected area networks. *People and Nature* 1, 331-346.

Denevan, W.M., 1992. The pristine myth: the landscape of the Americas in 1492. *Annals of the Association of American Geographers* 82, 369-385.

Drescher, M., Edwards, R.C., 2019. A systematic review of transparency in the methods of expert knowledge use. *Journal of Applied Ecology* 56, 436-449.

Dubois, N., Saulnier-Talbot, É., Mills, K., Gell, P., Battarbee, R., Bennion, H., . . . Valero-Garcés, B., 2018. First human impacts and responses of aquatic systems: A review of palaeolimnological records from around the world. *The Anthropocene Review* 5, 28-68.

Enright, N.J., Thomas, I., 2008. Pre-European fire regimes in Australian ecosystems. *Geography Compass* 2, 979-1011.

Ens, E.J., Pert, P., Clarke, P.A., Budden, M., Clubb, L., Doran, B., . . . Wason, S., 2015. Indigenous biocultural knowledge in ecosystem science and management: Review and insight from Australia. *Biological Conservation* 181, 133-149.

Eyre, T.J., Kelly, A.L., Neldner, V.J., 2006. Methodology for the establishment and survey of reference sites for BioCondition. Environmental Protection Agency, Queensland, Brisbane.

Faber-Langendoen, D., C. Hedge, M. Kost, S. Thomas, L. Smart, R. Smyth, . . . S. Menard, 2012. Assessment of wetland ecosystem condition across landscape regions: a multi-metric approach Part A. Ecological Integrity Assessment overview and field study in Michigan and Indiana. EPA/600/R-12/021a., p. 138. U.S. Environmental Protection Agency Office of Research and Development, Washington, DC.

Fensham, R., Fairfax, R., 1997. The use of the land survey record to reconstruct pre-European vegetation patterns in the Darling Downs, Queensland, Australia. *Journal of Biogeography* 24, 827-836.

Froyd, C.A., Willis, K.J., 2008. Emerging issues in biodiversity & conservation management: the need for a palaeoecological perspective. *Quaternary Science Reviews* 27, 1723-1732.

Gammage, B., 2011. *The Biggest Estate on Earth: How Aborigines made Australia*. Allen & Unwin, Crows Nest, New South Wales.

Gibbons, P., Briggs, S.V., Ayers, D., Seddon, J., Doyle, S., Cosier, P., . . . Roberts, K., 2009. An operational method to assess impacts of land clearing on terrestrial biodiversity. *Ecological Indicators* 9, 26-40.

Gibbons, P., Briggs, S.V., Ayers, D.A., Doyle, S., Seddon, J., McElhinny, C., . . . Doody, J.S., 2008. Rapidly quantifying reference conditions in modified landscapes. *Biological Conservation* 141, 2483-2493.



Harwood, T.D., Donohue, R.J., Williams, K.J., Ferrier, S., McVicar, T.R., Newell, G., White, M., 2016. Habitat condition assessment system: a new way to assess the condition of natural habitats for terrestrial biodiversity across whole regions using remote sensing data. *Methods in Ecology and Evolution* 7, 1050-1059.

Hector, A., Hautier, Y., Saner, P., Wacker, L., Bagchi, R., Joshi, J., . . . Loreau, M., 2010. General stabilizing effects of plant diversity on grassland productivity through population asynchrony and overyielding. *Ecology* 91, 2213-2220.

Heller, N.E., Hobbs, R.J., 2014. Development of a natural practice to adapt conservation goals to global change. *Conservation Biology* 28, 696-704.

Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C.K., . . . van de Bund, W., 2010. The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. *Science of The Total Environment* 408, 4007-4019.

Hessburg, P.F., Smith, B.G., Salter, R.B., 1999. Detecting change in forest spatial patterns from reference conditions. *Ecological Applications* 9, 1232-1252.

Hiers, J.K., Mitchell, R.J., Barnett, A., Walters, J.R., Mack, M., Williams, B., Sutter, R., 2012. The dynamic reference concept: measuring restoration success in a rapidly changing no-analogue future. *Ecological Restoration* 30, 27-36.

Hobbs, R.J., Higgs, E., Hall, C.M., Bridgewater, P., Chapin III, F.S., Ellis, E.C., . . . Yung, L., 2014. Managing the whole landscape: historical, hybrid, and novel ecosystems. *Frontiers in Ecology and the Environment* 12, 557-564.

Holling, C.S., 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4, 1-23.

Hooper, D.U., Chapin III, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., . . . Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecological Monographs* 75, 3-35.

Hughes, T.P., Barnes, M.L., Bellwood, D.R., Cinner, J.E., Cumming, G.S., Jackson, J.B.C., . . . Scheffer, M., 2017. Coral reefs in the Anthropocene. *Nature* 546, 82.

IPBES, 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services., eds S. Díaz, J. Settele, E. S., Brondízio E.S., H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, C.N. Zayas, p. 56, Bonn, Germany.

Jackson, S.T., 2013. Natural, potential and actual vegetation in North America. *Journal of Vegetation Science* 24, 772-776.

Joppa, L.N., Pfaff, A., 2009. High and far: biases in the location of protected areas. *PLoS ONE* 4, e8273.

Kahneman, D., 2011. *Thinking, fast and slow*. Macmillan, New York.

Keddy, P.A., Drummond, C.G., 1996. Ecological properties for the evaluation, management, and restoration of temperate deciduous forest ecosystems. *Ecological Applications* 6, 748-762.

Kimmerer, R.W., 2002. Weaving traditional ecological knowledge into biological education: a call to action. *Bioscience* 52, 432-438.

Kopf, R.K., Finlayson, C.M., Humphries, P., Sims, N.C., Hladysz, S., 2015. Anthropocene baselines: assessing change and managing biodiversity in human-dominated aquatic ecosystems. *Bioscience* 65, 798-811.

Kühn, I., Brandl, R., Klotz, S., 2004. The flora of German cities is naturally species rich. *Evolutionary ecology research* 6, 749-764.

Landres, P.B., Morgan, P., Swanson, F.J., 1999. Overview of the use of natural variability concepts in managing ecological systems. *Ecological Applications* 9, 1179-1188.

Lewis, S.L., Maslin, M.A., 2015. Defining the Anthropocene. *Nature* 519, 171.

Loidi, J., Fernández-González, F., 2012. Potential natural vegetation: reburying or reborning? *Journal of Vegetation Science* 23, 596-604.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., . . . Wardle, D.A., 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science* 294, 804-808.

Lotze, H.K., Worm, B., 2009. Historical baselines for large marine animals. *Trends in Ecology & Evolution* 24, 254-262.

Lu, C., Tian, H., 2014. Half-century nitrogen deposition increase across China: A gridded time-series data set for regional environmental assessments. *Atmospheric Environment* 97, 68-74.

Lunt, I.D., 2002. Grazed, burnt and cleared: how ecologists have studied century-scale vegetation changes in Australia. *Australian Journal of Botany* 50, 391-407.

McCann, K.S., 2000. The diversity-stability debate. *Nature* 405, 228-233.

McCarthy, M.A., Parris, K.M., Van Der Ree, R., McDonnell, M.J., Burgman, M.A., Williams, N.S.G., . . . Coates, T., 2004. The habitat hectares approach to vegetation assessment: an evaluation and suggestions for improvement. *Ecological Management & Restoration* 5, 24-27.

Mendel, L., 2002. The consequences for wilderness conservation in the development of the national park system in Tasmania, Australia. *Australian Geographical Studies* 40, 71-83.

Mihoub, J.-B., Henle, K., Titeux, N., Brotons, L., Brummitt, N.A., Schmeller, D.S., 2017. Setting temporal baselines for biodiversity: the limits of available monitoring data for capturing the full impact of anthropogenic pressures. *Scientific Reports* 7, 41591.

Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Synthesis*, Washington DC.

Molnár, Z., Kelemen, A., Kun, R., Máté, J., Sáfián, L., Provenza, F., . . . Vadász, C., 2020. Knowledge co-production with traditional herders on cattle grazing behaviour for better management of species-rich grasslands. *Journal of Applied Ecology* n/a.

Motzkin, G., Foster, D.R., 2002. Grasslands, heathlands and shrublands in coastal New England: historical interpretations and approaches to conservation. *Journal of Biogeography* 29, 1569-1590.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. *Nature* 403, 853-858.

Naeem, S., Li, S., 1997. Biodiversity enhances ecosystem reliability. *Nature* 390, 507-509.

Nagel, T.A., Levanic, T., Diaci, J., 2007. A dendroecological reconstruction of disturbance in an old-growth *Fagus-Abies* forest in Slovenia. *Annals of Forest Science* 64, 891-897.

Negret, P.J., Di-Marco, M., Sonter, L.J., Rhodes, J., Possingham, H.P., Maron, M., 2020. Effects of spatial autocorrelation and sampling design on estimates of protected area effectiveness. *Conservation Biology* n/a.

Oliver, I., Eldridge, D.J., Nadelny, C., Martin, W.K., 2014. What do site condition multi-metrics tell us about species biodiversity? *Ecological Indicators* 38, 262-271.

Pauly, D., 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends in Ecology & Evolution* 10, 430.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M., Kirschner, J., 2004. Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. *TAXON* 53, 131-143.

Rohwer, Y., Marris, E., 2016. Renaming restoration: conceptualizing and justifying the activity as a restoration of lost moral value rather than a return to a previous state. *Restoration Ecology* 24, 674-679.

Rose, D., Bell, D., Crook, D.A., 2016. Restoring habitat and cultural practice in Australia's oldest and largest traditional aquaculture system. *Reviews in Fish Biology and Fisheries* 26, 589-600.

Scholes, R.J., Biggs, R., 2005. A biodiversity intactness index. *Nature* 434, 45-49.

Schwartz, M.W., Brigham, C.A., Hoeksema, J.D., Lyons, K.G., Mills, M.H., van Mantgem, P.J., 2000. Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecologia* 122, 297-305.

Seabrook, L., McAlpine, C., Fensham, R., 2006. Cattle, crops and clearing: Regional drivers of landscape change in the Brigalow Belt, Queensland, Australia, 1840-2004. *Landscape and Urban Planning* 78, 373-385.

Seddon, A.W.R., Mackay, A.W., Baker, A.G., Birks, H.J.B., Breman, E., Buck, C.E., . . . Witkowski, A., 2014. Looking forward through the past: identification of 50 priority research questions in palaeoecology. *Journal of Ecology* 102, 256-267.

Seddon, J., Bourne, M., Murphy, D., Doyle, S., Briggs, S., 2011. Assessing vegetation condition in temperate montane grasslands. *Ecological Management & Restoration* 12, 141-144.

Shotbolt, L., Büker, P., Ashmore, M.R., 2007. Reconstructing temporal trends in heavy metal deposition: Assessing the value of herbarium moss samples. *Environmental Pollution* 147, 120-130.

Silcock, J.L., Piddocke, T.P., Fensham, R.J., 2013. Illuminating the dawn of pastoralism: evaluating the record of European explorers to inform landscape change. *Biological Conservation* 159, 321-331.

Sinclair, A.R.E., Mduma, S.A.R., Arcese, P., 2002. Protected areas as biodiversity benchmarks for human impact: agriculture and the Serengeti avifauna. *Proceedings of the Royal Society of London. Series B: Biological Sciences* 269, 2401-2405.

Sinclair, S.J., Griffioen, P., Duncan, D.H., Millett-Riley, J.E., White, M.D., 2015. Quantifying ecosystem quality by modeling multi-attribute expert opinion. *Ecological Applications* 25, 1463-1477.

Soga, M., Gaston, K.J., 2018. Shifting baseline syndrome: causes, consequences, and implications. *Frontiers in Ecology and the Environment* 16, 222-230.

Steffen, W., Grinevald, J., Crutzen, P., McNeill, J., 2011. The Anthropocene: conceptual and historical perspectives. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* 369, 842-867.

Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. *Ecological Applications* 16, 1267-1276.

Suding, K.N., 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. *Annual Review of Ecology, Evolution, and Systematics* 42, 465-487.

Swetnam, T.W., Allen, C.D., Betancourt, J.L., 1999. Applied historical ecology: using the past to manage for the future. *Ecological Applications* 9, 1189-1206.

Symstad, A.J., Jonas, J.L., 2014. Using natural range of variation to set decision thresholds: a case study for Great Plains grasslands, In *Application of Threshold Concepts in Natural Resource Decision Making*. ed. G.R. Guntenspergen, pp. 131-156. Springer New York, New York, NY.

Tilman, D., Reich, P.B., Isbell, F., 2012. Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. *Proceedings of the National Academy of Sciences* 109, 10394-10397.

Tilman, D., Reich, P.B., Knops, J., Wedin, D., Mielke, T., Lehman, C., 2001. Diversity and Productivity in a Long-Term Grassland Experiment. *Science* 294, 843-845.

Tilman, D., Reich, P.B., Knops, J.M.H., 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* 441, 629-632.

Tittensor, D.P., Mora, C., Jetz, W., Lotze, H.K., Ricard, D., Berghe, E.V., Worm, B., 2010. Global patterns and predictors of marine biodiversity across taxa. *Nature* 466, 1098-1101.

Turney, C.S.M., Palmer, J., Maslin, M.A., Hogg, A., Fogwill, C.J., Southon, J., . . . Hua, Q., 2018. Global Peak in Atmospheric Radiocarbon Provides a Potential Definition for the Onset of the Anthropocene Epoch in 1965. *Scientific Reports* 8, 3293.

Vihervaara, P., D' Amato, D., Forsius, M., Angelstam, P., Baessler, C., Balvanera, P., . . . Zacharias, S., 2013. Using long-term ecosystem service and biodiversity data to study the impacts and adaptation options in response to climate change: insights from the global ILTER sites network. *Current Opinion in Environmental Sustainability* 5, 53-66.

Villnäs, A., Norkko, A., 2011. Benthic diversity gradients and shifting baselines: implications for assessing environmental status. *Ecological Applications* 21, 2172-2186.

Watson, J.E.M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., . . . Lindenmayer, D., 2018. The exceptional value of intact forest ecosystems. *Nature Ecology & Evolution* 2, 599-610.

White, M.A., Mladenoff, D.J., 1994. Old-growth forest landscape transitions from pre-European settlement to present. *Landscape Ecology* 9, 191-205.

Whittaker, R.H., 1967. Gradient analysis of vegetation. *Biological Reviews* 42, 207-264.

Wolman, A.G., 2006. Measurement and meaningfulness in conservation science. *Conservation Biology* 20, 1626-1634.

Yachi, S., Loreau, M., 1999. Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis. *Proceedings of the National Academy of Sciences* 96, 1463-1468.

Yen, J.D.L., Dorrough, J., Oliver, I., Somerville, M., McNellie, M.J., Watson, C.J., Vesk, P.A., 2019. Modeling biodiversity benchmarks in variable environments. *Ecological Applications* 29, 1-16.

Zerbe, S., 1998. Potential natural vegetation: validity and applicability in landscape planning and nature conservation. *Applied Vegetation Science* 1, 165-172.

Zhang, K., Yang, X., Xu, M., Lin, Q., Kattel, G., Shen, J., 2018. Confronting challenges of managing degraded lake ecosystems in the Anthropocene, exemplified from the Yangtze River Basin in China. *Anthropocene* 24, 30-39.

Zu Ermgassen, P.S.E., Spalding, M.D., Blake, B., Coen, L.D., Dumbauld, B., Geiger, S., . . . Brumbaugh, R., 2012. Historical ecology with real numbers: past and present extent and biomass of an imperilled estuarine habitat. *Proceedings of the Royal Society B: Biological Sciences* 279, 3393-3400.

Zweig, C.L., Kitchens, W.M., 2010. The Semiglades: the collision of restoration, social values, and the ecosystem concept. *Restoration Ecology* 18, 138-142.