Anthropocene Baselines: Assessing Change and Managing Biodiversity in Human-Dominated Aquatic Ecosystems

R. KELLER KOPF, C. MAX FINLAYSON, PAUL HUMPHRIES, NEIL C. SIMS, AND SALLY HLADYZ

Global ecosystems have shifted from historical conditions, but it is unclear from what baselines change should be assessed. Scientists and managers have increasingly accepted the impossibility of returning ecosystems to a "pristine" state; however, historical conditions remain the cornerstone for restoration and management. We explore the rationale behind the application of historical baselines to ecosystem management and propose Anthropocene *baselines as a concept to provide an improved basis for the management of human-dominated ecosystems. The* Anthropocene *baselines concept emphasizes the conservation value of the remnants of historical ecosystems but confronts the reality that many ecosystems cannot—or will not—be restored to historical ranges of variability. In order to prevent further unwanted changes to biodiversity and ecosystem services, we suggest that the management of human-dominated ecosystems must move beyond historical constraints toward new points of reference dictated by social–ecological sustainability.*

Keywords: river restoration, regime shift, resilience, alternative stable state, novel ecosystem

World governments have agreed to targets that aim to alleviate the global biodiversity crisis and to improve the livelihood and long-term sustainability of humans. The high degree of human dependence on biodiversity is formally recognized by the United Nations, through the Convention on Biological Diversity, the Millennium Development Goals, and the Aichi Strategic Plan for Biodiversity 2011–2020 (United Nations 2010). Per unit of habitat area, freshwater ecosystems are among the most diverse and human-altered environments on the planet (box 1, table 1). Extensive physical, chemical, and biotic transformations in freshwater ecosystems (Millennium Ecosystem Assessment 2005, Carpenter et al. 2011) have demonstrated that many human-induced changes are practically irreversible (Carpenter et al. 1999, Downing et al. 2012, Hobbs WO et al. 2012, Moyle 2014).

A rapidly expanding area of research aims to identify and predict the incidence, location, timing and effects of regime shifts, often referred to as *ecosystem collapses* or *state changes* (Scheffer et al. 2001, 2009, Andersen et al. 2009, Barnosky et al. 2012). The *regime-shift* concept holds that if a critical threshold is passed in a highly connected system (Scheffer et al. 2009), such as a river network (Dent et al. 2002), a new self-reinforcing state may occur. These shifts can arise rapidly or inconspicuously over decades and centuries (Hughes TP et al. 2013). Evidence supporting the occurrence of state changes in lake ecosystems is well established (Carpenter et al. 1999, Hobbs WO et al. 2012), but much less is known about alternative stable states in river and floodplain ecosystems (Dent et al. 2002, Heffernan 2008, Ibanez et al. 2012).

Historical baselines, in the relative absence of modern humans, are the conceptual benchmark for biodiversity assessment and management, but it is reasonable to ask whether past conditions remain relevant reference points for the restoration and management of contemporary human-altered systems. The concept of *novel ecosystems* (Hobbs RJ et al. 2013) posits that human-induced changes to species assemblages and ecosystem functioning have permanently transformed many ecosystems. This concept implies that ecosystems have arrived at a new point of reference. Therefore, using historic baselines for the conservation and management of certain human-dominated ecosystems may be inappropriate (Suding 2011). We suggest that this gap is especially acute for highly altered freshwater ecosystems.

Here, we examine baselines used to assess the state of global freshwater biodiversity and explore the rationale

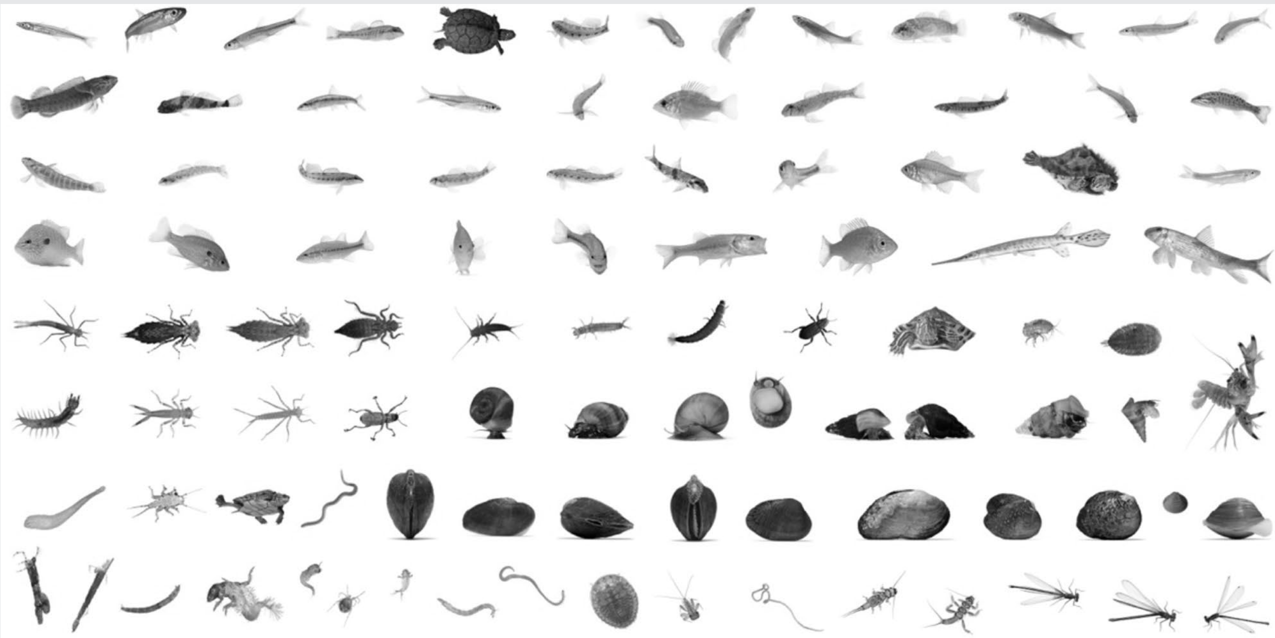
BioScience 65: 798–811.
doi:10.1093/biosci/biv092 Advance Access publication 22 July 2015

Box 1. Freshwater ecosystems: Global hotspots of biodiversity.

Freshwater habitats make up less than 2% of the Earth's surface but are home to approximately 10% of all described species of fungi, plants, invertebrates, and vertebrates (table 1). The freshwater realm supports more than 8 and 38 times (table 1) more vertebrate species per unit area than land and sea, respectively. In addition to high species richness and endemism, estimates from North America suggest that freshwater biota have extinction rates approximately five times greater than terrestrial and marine animals (Ricciardi and Rasmussen 1999). Extinction rates of freshwater fishes are about 203 times the background extinction rate prior to the twentieth century (Burkhead 2012), and those of freshwater mussels and snails are at more than 400 times background levels (Ricciardi and Rasmussen 1999).

The best estimates suggest that 50%–65% of all freshwater ecosystems have been lost or moderately high to severely altered (table 1). Well over 70% of floodplains in the Murray-Darling Basin, the Mississippi River, parts of Asia, and all of Europe have been converted to cropland or urban development (Tockner et al. 2008). Large tropical river systems, such as the Amazon, Congo, and Mekong, support a majority of the world's vertebrate species (Tisseuil et al. 2013), and these environments are under threat because of forestry, agriculture, and water resource development (Vorosmarty et al. 2010).

It would be an exaggeration to suggest that the freshwater realm is a biodiversity hotspot in its entirety, but particular wetlands and river systems are certainly comparable to marine and terrestrial centers of diversity and anthropogenic modification.



Eighty-one species of fish, reptiles, crustaceans, mollusks, nematodes, and other invertebrates visible to the naked eye within one cubic foot of the Duck River, Tennessee, over a 24-hour period. Photograph: Reprinted with permission from Liittschwager (2012).

behind the application of historical baselines in ecosystem assessment. We propose *Anthropocene baselines* (ABs) as a concept to recognize human-induced shifts in the rehabilitation capacity of many ecosystems. ABs represent an ecological and theoretical shift away from the constraints of historical references and provide a basis to begin acknowledging new points of reference for managing biodiversity and assessing change in human-dominated ecosystems.

Traditional baselines

Biodiversity assessment and management require standards from which change can be measured. Because baselines are typically used to assess the effects of humans on ecosystems, a point of reference in the absence of human activity is a logical standard. In assessments of aquatic biodiversity, such standards are usually referred to as *reference conditions* (Hughes RM et al. 1986, Stoddard et al. 2006) or *baselines* (Pauly 1995, Humphries and Winemiller 2009). Baselines in the absence of European human disturbances have been essential to ecology and natural resource management since at least the early 1900s (Adams 1913), and they remain fundamental to the way in which we interpret degradation and the effectiveness of conservation and restoration activities.

Table 1. Global habitat area and catalogued extant species area summaries comparing freshwater, marine, and terrestrial realms.

Global biodiversity attribute	Freshwater	Marine	Terrestrial
Surface area (in square kilometers [km^2])	11,887,000	361,300,000	131,900,000
Percentage earth surface area	2	72	26
Percentage estimate of native habitat lost or moderately high to severely altered	50–65	41	21.8–45.3
Fungi	1960	1097	41,311
Percentage of global total	4	3	93
Global species per 1000 km^2	0.17	0.003	0.31
Vascular plants and related algae	9976	8600	215,644
Percentage of global total	4	4	92
Global species per 1000 km^2	0.84	0.02	1.64
Invertebrate Animalia	107,295	155,128	803,562
Percentage of global total	10	15	75
Global species per 1000 km^2	9.03	0.43	6.10
Vertebrate Animalia	18,235	15,954	24,342
Percentage of global total	31	27	42
Global species per 1000 km^2	1.53	0.04	0.19
Total species	137,466	180,779	1,084,859
Percentage of global total	10	13	77
Global species per 1000 km^2	11.56	0.50	8.22

Note: See supplementary table S1 for calculations and references.

Hominids have been modifying nature to a significant extent for millennia, but industrial-era technology, starting around 1800, resulted in an exponential rise in anthropogenic modification and use of natural resources (Steffen et al. 2007). The rise in anthropogenic pressures during the industrial era largely initiated the Anthropocene (Steffen et al. 2007), and these changes occurred at different times around the world (Lewis and Maslin 2015). Baselines used to assess changes in freshwater environments typically apply a reference year between 1750 or 1800. Over the following centuries, the state of biodiversity in most freshwater ecosystems moved outside of preindustrial-era ranges of variability (Humphries and Winemiller 2009, Carpenter et al. 2011). Some systems, such as rivers in Scandinavian countries, were disturbed to a lesser extent than others, such as those in agricultural regions of the midwestern United States, Europe, and Australia (Stoddard et al. 2006).

Least disturbed reference sites remain the most common baseline used to measure restoration and management outcomes in the assessment of river and stream biodiversity. Stoddard and colleagues (2006) defined the *reference condition* as a best estimate of the natural structure and function of biota and ecosystems in the absence of human disturbance. This approach compares indicator data from an array of sampling sites to similar but less disturbed baseline rivers or streams. Least disturbed sites are held to represent the best real-world examples of conditions in the absence of humans, but virtually all have been influenced by anthropogenic pressures. The reference-condition approach also generally fails to recognize spatial and temporal variability in quality and availability of least disturbed sites. Therefore, rivers in many parts of the world that are classified as *least disturbed* baselines are, in fact, highly altered by human activity.

The global assessment of biodiversity, undertaken by the Convention on Biological Diversity (Butchart et al. 2010), used an arbitrary and fairly recent year, 1970, as the baseline. Ramsar wetlands use a similar approach to assess *ecological character* (Pittock et al. 2010) when a site is first listed, and then trends in biodiversity are assessed against this contemporary baseline every six years. However, there are no international standards or guidelines for defining baselines. Without standards, Ramsar ecological-character assessments can be modified, effectively shifting the baseline to suit political or other agendas.

An example in which contemporary baselines diverge greatly from preindustrial era baseline conditions is the mouth of the Murray River, Australia (box 2). Hypersaline conditions, in the Coorong, at the time of the Ramsar listing in 1985 were assumed to be the natural baseline conditions. Management actions were therefore directed toward maintaining the 1985 hypersaline biota that survived in these harsh conditions. Paleolimnological evidence subsequently revealed that the salinity in the system had increased dramatically with European settlement, especially with the construction of large dams and barrages (Dick et al. 2011). The biota, which are now established provide new and occasionally desirable ecosystem functions, such as food for waterbirds, but some of these species did not exist in the system prior to European settlement.

Box 2. Mouth of the Murray-Darling River System, Australia: An Anthropocene baseline.

The Coorong, Lower Lakes, and Murray Mouth (CLLMM) were listed as a Ramsar wetland of international importance in 1985, largely as a consequence of their significance as a feeding ground for migratory waterbirds. An assessment of the "ecological character" of the CLLMM in 1985 recorded the Coorong as hypersaline and the Lower Lakes as permanently freshwater (Kampf and Bell 2014). These conditions served as the baseline for management, including restoration. However, paleolimnological and historical evidence has revealed that the contemporary 1985 baseline conditions differ greatly from those that occurred prior to European settlement (Dick et al. 2011, Reeves et al. 2014).

Prior to European settlement, water from Australia's largest river system, the Murray-Darling, flowed into Lakes Alexandrina and Albert (the Lower Lakes) and through an estuarine lagoon (the Coorong) before exiting to the Southern Ocean. The mixing of river and tidal water created a mosaic of connected and dynamic environments ranging from freshwater–brackish lakes and wetlands to estuarine marshes, floodplains, mud, sand, and salt flats (Kampf and Bell 2014). These conditions changed because of a variety of anthropogenic modifications, including the construction of large dams, locks, and barrages, mostly completed by the mid-1940s. These changes and others resulted in parts of the Coorong shifting to hypersaline conditions and the Lower Lakes being isolated from the incoming tides and shifting to freshwater conditions, except during extreme drought. Native diadromous fishes, such as Congolli (*Pseudaphritis urvillii*), have declined in lakes, whereas invasive generalists, including common carp (*Cyprinus carpio*), now dominate the freshwater fish assemblage (Wedderburn et al. 2014).

Mouth of the Murray-Darling 1932–1939. Photograph: Harold Beaney, courtesy of the State Library of South Australia.

Mouth of the Murray-Darling, 2008. Photograph: Courtesy of the Australian heritage photographic library, Department of the Environment.

Given the severity of the changes and the near impossibility of reversing these modifications, restoration to a pre-European baseline is not considered feasible and may be counterproductive to maintaining biodiversity. For example, a species of aquatic plant, *Ruppia tuberosa*, which never existed in parts of the system before European settlement (Dick et al. 2011), is now the primary source of food supporting several migratory waterbird populations. We question whether adherence to historical baseline conditions provides the most meaningful point of reference to guide biodiversity management in this human-dominated ecosystem. Instead, new baseline conditions following preindustrial era changes may represent a more appropriate point of reference.

We refer to the contemporary human-modified baseline of the CLLMM and other highly modified environments as an *Anthropocene baseline*. This recognizes that humans are the dominant force determining ecological processes and biodiversity. These systems are not expected to return to historical ranges of variability because of the legacy of past anthropogenic change and continuing human pressures. We therefore suggest that rehabilitation efforts need to be more flexible and directed at working within the social–ecological system to maintain desirable attributes of biodiversity irrespective of historical precedence.

Shifting baselines and restoration paradigms

Reliance on contemporary data in evaluating biodiversity changes can entrain a shifting baseline syndrome, in which historical points of reference are forgotten with each passing generation (Pauly 1995). Forgetting historical references can limit our perspective on the restoration potential of ecosystems (see Jackson et al. 2001, 2011) and can increase the risks of the unexpected collapse of ecosystem services, such as those provided by fisheries or fresh water (Dearing et al. 2012).

Large, persistent, and often unexpected changes may result in ecosystem regime shifts (Scheffer et al. 2001, 2009, Biggs et al. 2012, Hughes TP et al. 2013) that alter restoration pathways and shift systems into a new state of equilibrium.

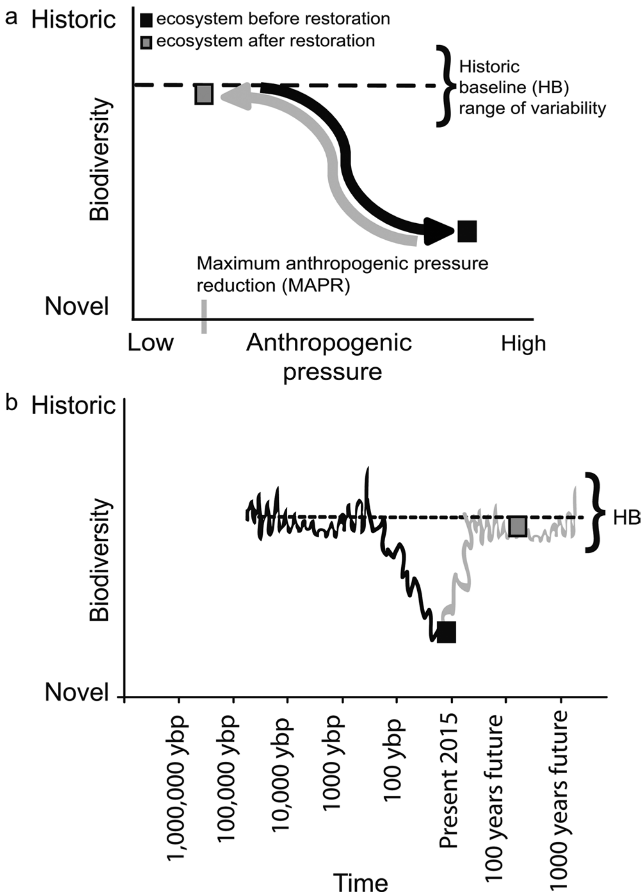


Figure 1. Current river and wetland restoration paradigm. The trajectory of biodiversity decline (the solid black line and arrow; Davies and Jackson 2006) is assumed to match the trajectory of recovery (solid grey line and arrow). Panel (a) illustrates the assumed mechanistic relationship between increasing anthropogenic pressure and decreasing biodiversity followed by proportional restoration to the historic baseline (HB; the black dashed line) range of variability. Panel (b) illustrates the same relationship as shown in (a) but over a hypothetical time period encompassing the onset of the Anthropocene, c. 1610 or 1964 (Lewis and Maslin 2015).

It is now widely recognized that disturbance is a normal part of ecosystem functioning and that therefore, most systems, including large rivers (Strayer et al. 2014) and small streams (Moyle 2014), do not exist in a permanent state of equilibrium (Chapin et al. 2009). Therefore, the concept of a static historical baseline as a universal point of reference has resulted in management expectations that do not account for ecological variability.

It remains difficult to detect whether most systems have actually entered a new and stable state. However, a number of large and difficult-to-reverse ecosystem changes have been documented in the scientific literature, including transitions from oligotrophic to eutrophic lakes (Carpenter et al. 1999, Hobbs WO et al. 2012), the desertification of the Aral Sea (Micklin 2010), and the collapse and reorganization of Lake Victoria's fish assemblage following the introduction of Nile perch (*Lates niloticus*; Downing et al. 2012). Here, we highlight the difficulty of reversing changes affected by dams, flow alteration, invasive species, and freshwater extraction in regulated river systems (box 2). Recent evidence suggests that less obvious regime shifts may be widespread and insidious, particularly when changes unfold slowly over long periods of time (Hughes TP et al. 2013).

Indeed, most river–floodplain systems have shifted from historical baselines over long periods of time. Even disregarding land-use modification, climate change, invasive species, habitat removal, or early commercial fishing (see Humphries and Winemiller 2009, Carpenter et al. 2011), the alteration of natural flow regimes by large dams alone has fundamentally shifted ecosystem functions and the biota of rivers and wetlands worldwide (Poff et al. 1997). There is growing discussion surrounding the changing nature of human-modified rivers and floodplain systems and how the restoration and management of these systems should be approached (Beechie et al. 2010, Moreno-Mateos et al. 2012, Moyle 2014, Strayer et al. 2014).

The current river and wetland restoration paradigm presumes that the historic assemblages and processes of rivers and wetlands can be restored by incrementally reversing anthropogenic pressure (e.g., Davies and Jackson 2006, Stoddard et al. 2006). Based on this assumption, the empirical relationship between past ecological responses and increasing anthropogenic pressure (figure 1a) is used to guide restoration activities that aim to reverse degradation. For example, the relationship between declining biodiversity and increasing natural flow regime disturbance provides the foundation for predicting the quantity and timing of water needed to be returned to rivers for restoration (Poff et al. 1997, 2009, Arthington et al. 2006).

In effect, management activities aim to return systems toward a state that occurred in the past (figure 1b) by restoring processes and habitat. The current river restoration paradigm acknowledges that many degraded ecosystems will not be restored to a historic baseline (Beechie et al. 2010) or a minimally disturbed condition (*sensu* Stoddard et al. 2006), but they remain based on the expectation that contemporary human-modified ecosystems will respond to restoration in the same manner as more pristine rivers. If, for example, the maximum reduction in anthropogenic pressure (e.g., natural flow regime disturbance) were reversed to historic levels, biodiversity would be expected to return to historic ranges of variability (figure 1a, 1b). Past ecosystems, however, may no longer be a useful guide to predict current or future responses in human-modified river ecosystems.

The current paradigm in river and wetland restoration presumes that the trajectory of recovery will match the trajectory of decline (figure 1a). This approach fails to account for aspects of biodiversity and ecosystem functioning that have been irreversibly altered from past conditions. For example, invasive species, climate change, and the persistence of

Table 2. Criteria for distinguishing Anthropocene and historical baselines, including links to the novel ecosystem concept and fundamental differences in management intervention approach.

Baseline	Criteria	Ecosystem type[c]	Principal management intervention	Potential examples
Anthropocene baseline	Ecosystem, or biodiversity attribute[a], will not be restored to the historical range of variability[b] because of socioeconomic (type I) constraints on reducing anthropogenic pressures or internal ecological (type II) constraints.	Novel	Ecosystem stewardship[d] limited to meaningful interventions[e] that foster social–ecological sustainability and maintain desirable attributes of biodiversity, irrespective of historical precedence.	Reservoirs, heavily regulated rivers and streams, associated wetlands and lakes in urban or agricultural landscapes and floodplains, ecosystems heavily altered by invasive species, pollution, physical modification and harvesting, or low resistance to climate change.
Anthropocene baseline (without restoration and conservation) or historical baseline (with restoration and conservation)	Restoration of ecosystem, or biodiversity attribute[a], to the historical baseline range of variability[b] may be feasible but expected to arrive at an Anthropocene baseline without further management intervention.	Hybrid	Classic restoration and conservation[f] to reverse degradation back to historical ranges of variability,or maintain *status quo* and revert to Anthropocene baseline.	Largely free-flowing rivers and streams, associated wetlands and lakes in areas with low to moderate levels of nonnative species invasion, land-use modification, pollution, physical habitat modification, harvesting and moderate to high resistance to climate change.
Historical baseline	Ecosystem, or biodiversity attribute[a], is within historical baseline range of variability[b], or is expected to return to the historical range of variability if increases in anthropogenic pressures are limited.	Historical	Classic conservation[f]	Free-flowing river systems and associated wetlands and lakes within wilderness landscapes where humans are not the dominant ecological force despite unavoidable impacts from industrialized society, climate change and invasive species.

[a]Noss (1990).
[b]Wiens et al. (2012).
[c]Hobbs et al. (2013).
[d]Chapin et al. (2009).
[e]Hobbs et al. (2011).
[f]Young (2000).

modified landscapes all influence restoration potential. It is unreasonable to expect that all human-modified ecosystems will respond similarly to restoration and management activities regardless of the degree of alteration.

The degradation of freshwater, marine, and terrestrial environments, coupled with unexpectedly slow or no recovery (Suding 2011, Downing et al. 2012, Hobbs WO et al. 2012, Neubauer et al. 2013), suggests that human-induced alterations of some environments and populations can and do limit rehabilitation potential. Small-scale restoration projects can have positive short-term responses, which are enticing, but numerous large-scale river and wetland restoration efforts around the world have met with limited success when compared to historical references (Palmer et al. 2010, Suding 2011, Moreno-Mateos et al. 2012). Instead, “partial restoration” is the most commonly reported result following river restoration (Beechie et al. 2010).

Anthropocene baselines (ABs)

Scientists and managers accept the impossibility of returning ecosystems to a “pristine” state; however, historical baselines remain the conceptual cornerstone of managing and restoring human-modified systems. We argue that expectations set by historical references alone will not accurately predict the dynamics of how contemporary human-modified systems will respond to management, restoration, and future environmental changes.

We propose *Anthropocene baselines* (ABs) as a concept to represent an ecological and theoretical shift from a fixed preindustrial-era reference condition to a dynamic point of reference for human-dominated ecosystems. We call these *Anthropocene baselines* because they represent a new rehabilitation potential for most ecosystems, catalyzed by humans following the industrial era (circa 1800–1945, Steffen et al. 2007) and the subsequent acceleration of anthropogenic pressures. The precise year marking the onset of ABs will vary according to regional differences in human activity, but 1610 or 1964 (Lewis and Maslin 2015) have been proposed as global standards for defining the beginning of a new geological epoch.

The criterion for the classification of an AB (table 2) is that the ecosystem will not be restored to the historical baseline range of variability (see Wiens et al. 2012). ABs do not apply to all ecosystems modified by humans or invasive species, but many freshwater, marine, and terrestrial ecosystems would likely fit the criterion. ABs may arise because of socioeconomic constraints which limit reductions in anthropogenic pressure or because of internal ecological constraints affected by novel invasive species or the legacy effects of past alteration (table 2). Therefore, we propose that ABs apply when the best attainable rehabilitation potential is outside of the preAnthropocene range of variability. Unlike historic baselines, the AB concept integrates the effects of past and present human disturbance to acknowledge that certain ecosystems have arrived at a new point of reference under conditions of the Anthropocene (e.g., box 2).

Anthropocene baselines at multiple levels of ecological organization. Ecosystem state changes and regime shifts are most easily identified using long time-series monitoring data (e.g., Strayer et al. 2014) and have been reported across a range of terrestrial and aquatic environments (Scheffer et al. 2001, 2009, Andersen et al. 2009). A ball and cup diagram is often used to illustrate how a state shift occurs, in which the ball (ecosystem) is pushed into a new cup (state basin of attraction). The same force (anthropogenic pressure) in the opposite direction (restoration) will not return the ball to its former state basin (Beisner et al. 2003). State changes and regime shifts (Scheffer et al. 2001, Andersen et al. 2009) caused by humans represent ABs at the ecosystem level of organization. As more attributes of biodiversity or ecological drivers (e.g., species traits, population productivity, climate, river flow regime) within an ecosystem shift from historical ranges of variability, we hypothesize that entire systems become increasingly likely to experience a regime shift and therefore arrive at an AB.

The human-induced evolution of species' traits can result in ABs. Examples include resistance to pesticides, herbicides, and antibiotics, as well as human-induced alterations in growth rate, morphology, distribution, and the timing of reproduction (Palumbi 2001, Allendorf and Hard 2009). For example, in the polluted Hudson River, tomcod (*Microgadus tomcod*) are now approximately 100 times more resistant to cancerous hydrocarbons than the same species of fish in less contaminated waters nearby (Strayer et al. 2014). Slow or no recovery of traits, following the cessation of the anthropogenic pressure which caused the change, appears to be a common feature (Allendorf and Hard 2009).

Human-induced reductions in population productivity, including Allee effects (Courchamp et al. 2008), contribute to ABs. In contrast to the classical view of negative density-dependent population dynamics, some species become less resilient with decreasing population size (Courchamp et al. 2008). This phenomenon is exemplified by some lacustrine fish communities of North America, in which adults consume competitors or the predators of their own offspring (Walters and Kitchell 2001). Consequently, negative feedback occurs, in which recruitment of these lacustrine fishes declines when adults are harvested. In extreme cases, the carrying capacity of populations may remain in a depressed state that would be classifiable as an AB. Many commercially important fish populations are expected to recover if fishing pressure is reduced, but Allee effects and altered resilience can prevent the recovery of severely overexploited stocks (Neubauer et al. 2013).

Anthropocene Baselines may be reinforced by internal ecosystem changes and feedback, human activity, or by a combination of both. For example, eutrophication in lakes may be initiated by human-induced increases in nutrient inputs, but a turbid, eutrophic state may persist because of a lower abundance of large piscivorous fishes (Hobbs WO et al. 2012). Therefore, reducing nutrient inputs alone will not catalyze the same response as historic conditions. Changes in other ecosystems may follow a simple gradient by which degradation can be reversed (Carpenter et al. 1999) by incrementally reducing anthropogenic pressures (figure 2a). Even in these systems, however, socioeconomic constraints often limit restoration to historical ranges of variability (figure 2b). If the rehabilitation potential of an ecosystem lies within the historical range of variability, then it is not an AB. Distinguishing whether ABs are reinforced by ecological constraints, human activities, or both will be important in prioritizing management actions to address underlying problems. We highlight two different types of ABs, although both types may apply in many circumstances.

Type I Anthropocene baselines: Restoration is limited by socioeconomic constraints. Type I ABs arise because of socioeconomic limitations on reducing anthropogenic pressure (figure 2b) rather than an ecological state change or regime shift. Type I ABs are considered to have crossed a noncatastrophic threshold (Scheffer et al. 2009); therefore, changes are considered to be reversible given adequate socioeconomic impetus.

An example of a type I AB is the Aral Sea. This system experienced a twentyfold increase in salinity and an 88% reduction in surface area following intense river water extraction since the 1960s (Micklin 2010). Rehabilitation efforts of increasing freshwater inflows in parts of the system have been highly successful at diluting salinity, but socioeconomic limitations on returning adequate supplies of freshwater prevent full restoration to pre-1960 levels (total surface area and salinity) (Micklin 2010). Another example of a type I AB is attempting to restore the natural flow regime in a highly regulated river system. Removal of all dams, restoring river channel and floodplain structure and connectivity, and eliminating water extraction *may* altogether restore the natural flow regime back to historical ranges of variability. However, socioeconomic constraints related to food production, the livelihood of people in rural communities, and the monetary value of agriculture ultimately prevent the full restoration of the natural flow regime to historical ranges of variability in most river systems (see Beechie et al. 2010).

Because the flow regime is the principal abiotic driver supporting river ecosystems (Poff et al. 1997), many biotic attributes of biodiversity, such as population biomass and native species richness in highly regulated rivers, will not be restored to historical ranges of variability. Consequently, highly regulated rivers and streams should be carefully evaluated to assess whether historical baselines or ABs are

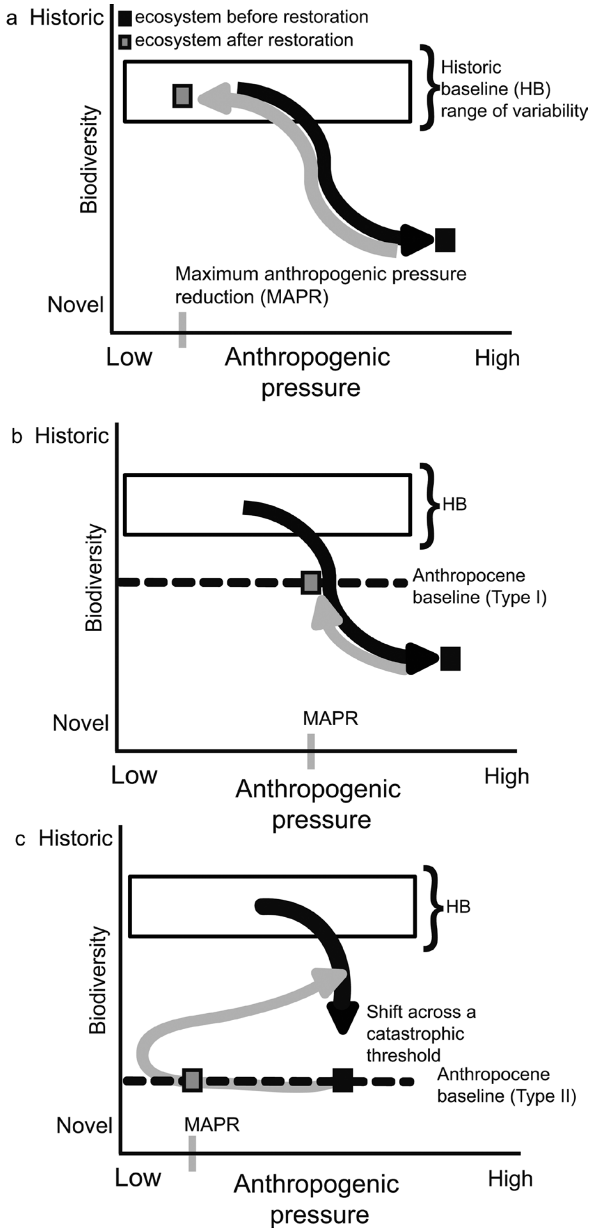


Figure 2. Current river and wetland restoration paradigm (a) where the trajectory of biodiversity decline (solid black line and arrow) is assumed to match the trajectory of recovery (solid grey line and arrow). Anthropocene Baselines (ABs; black dashed lines) arise when the restoration potential is outside of the historical baseline (HB; the black box) range of variability. Type I ABs (b) arise because of socioeconomic limits on reducing anthropogenic pressure; type II ABs (c) arise when the maximum anthropogenic pressure reduction (MAPR; e.g., restoration of the natural flow regime) is expected to return the system to an HB but internal ecological constraints and feedback mechanisms (e.g., invasive species reproduction) alter the restoration trajectory. Type II ABs represent a state change across a catastrophic threshold sensu Scheffer et al. (2001, 2009), whereas type I ABs may arise without a state change.

the most appropriate guide for management. However, it is important to highlight that some attributes of highly altered ecosystems may remain within historical ranges of variability despite major changes to the system.

Type II Anthropocene baselines: Restoration is limited by ecological constraints. In systems that have been modified to extreme levels, there is compelling evidence (Carpenter et al. 1999, Scheffer et al. 2001, 2009, Dent et al. 2002, Suding et al. 2004, Hughes TP et al. 2013) to suggest that recovery does not always follow the trajectory predicted by reversing anthropogenic pressure (figure 2c). Type II ABs occur when systems are considered to have experienced a state change or regime shift by crossing a catastrophic threshold (Scheffer et al. 2001, 2009). The presence of a regime shift in long time-series monitoring data can be tested for statistically using nonlinear modeling approaches (Andersen et al. 2009).

Many of the best examples of human-induced state changes and, therefore, type II ABs come from freshwater lake studies describing shifts between clear oligotrophic and turbid eutrophic states (Carpenter et al. 1999, Scheffer et al. 2001, Hobbs RJ et al. 2011). When nutrient concentrations from agricultural run-off exceed a threshold which aquatic plants can absorb, a dense growth of algae blooms that reduces light penetration in the water column. Reduced light levels lead to the death of the aquatic plants. This, in turn, suspends soil in the water column, leading to increased turbidity and a release of more nutrients, which creates a feedback loop perpetuating further algal growth. Type II ABs occur when anthropogenic pressure (e.g., nutrient input) is reversed far enough to attain a historic baseline (e.g., oligotrophic state), but the system remains in a state (e.g., eutrophic state) outside of the historical range of variability (figure 2c). In severe cases, lake eutrophication (Carpenter et al. 1999, Hobbs WO et al. 2012) may be considered irreversible. Importantly, however, not all state changes are irreversible (see Downing et al. 2012).

Most freshwater examples of human-induced regime shifts come from lake ecosystems that provide closed and stable systems, which are well suited for experimental manipulation. Much less is known about alternative states in rivers, which are more connected and dynamic than lake ecosystems. River ecosystems around the world have experienced large and rapid changes due to human activities (e.g., Humphries and Winemiller 2009, Strayer et al. 2014), but it remains an open research question whether these activities have shifted rivers into a new state during the Anthropocene. Indeed, there is growing discussion around novel ecosystem concepts in managing flowing waters, and many systems are not expected to return to historical conditions (Moyle 2014). Rivers undergo state changes associated with flooding and drought, river channel geomorphology, riparian zone vegetation (Dent et al. 2002), and even shifts between clear oligotrophic and turbid eutrophic states (Ibanez et al. 2012). Invasive species have been shown to limit rehabilitation potential in rivers, where the natural flow regime has been restored to near historic levels via dam removal (Marks et al. 2009). These internal factors, which limit ecosystem rehabilitation and therefore maintain type II ABs, are more broadly referred to in the regime shift literature as *self-reinforcing feedback loops* (Biggs et al. 2012).

Baselines for social–ecological sustainability. Most biodiversity monitoring programs are based on the concept of a *historic baseline*, the condition in the relative absence of modern humans. This concept is difficult to justify when humans and biodiversity are viewed as part of a single social–ecological system (Chapin et al. 2009). Anthropocene baselines may be useful reference points for evaluating social–ecological sustainability but will often be quite degraded from a historic point of view.

Unlike the shifting baseline syndrome (Pauly 1995), in which historic references have been forgotten by people, ABs represent anthropogenic changes to the rehabilitation potential of ecosystems that exist today. The novel-ecosystem (Hobbs RJ et al. 2013), alternative stable state (Scheffer et al. 2001, 2009), and regime-shift (Biggs et al. 2012) concepts concede that certain changes are practically irreversible, but none of these approaches explicitly addresses the need to redefine baselines for the management of human-dominated systems.

Achieving social–ecological sustainability, whether it be maintaining ABs or preventing further ABs from emerging, will ultimately require the cessation of infinite economic growth, supported by the increasing consumption of nonrenewable natural resources. The risks of not incorporating anthropogenic change into estimating rehabilitation potential include (a) overly optimistic management expectations that lead to unsustainable rates of natural resource use and (b) wasted resources chasing unattainable restoration goals. Conversely, by incorporating anthropogenic change into points of reference, there is a danger that rehabilitation targets will be dominated by greed at the cost of biodiversity. We suggest, therefore, that ABs must be based on the maximum achievable level of reduction in anthropogenic pressure and a balance between social and ecological sustainability (see Chapin et al. 2009).

Anthropocene baseline identification. The first step in identifying whether ABs apply is to determine whether the candidate ecosystem is within historical ranges of variability (figure 3). We envisage two broad ways in which ABs could be identified: *post hoc* and *a priori*. A precautionary approach to evaluate whether ABs apply would implement full restoration and conservation actions and then evaluate *post hoc* whether biodiversity is within historical ranges of variability. If not, then ABs should be quantified.

A *post hoc* AB identification process would be based on time-series monitoring data, evaluating whether the system recovers to historical ranges of variability following management actions. The AB becomes apparent when, in attempting to achieve those objectives, restoration to historical ranges of variability is not attained. For example, river habitat restoration projects are occurring across most of eastern North America to help restore Pacific salmonid populations. Extensive monitoring of ecosystems and populations prior to restoration activities provides a sound footing on which to evaluate potential restoration responses, but because of the confounding effects of invasive species and other food-web limitations on carrying capacity (Naiman et al. 2012), it remains uncertain whether populations will return to historical ranges of variability.

An *a priori* AB analysis would include an initial pilot to model: (a) the maximum achievable level of reduction in anthropogenic pressure, (b) the rehabilitation potential of the ecosystem being restored in relation to the maximum achievable level of reduction in anthropogenic pressure, and (c) the historic baseline reference. Although conducting an *a priori* AB analysis may add to the initial cost of restoration and management activities, this method may save time and money in the long term by identifying key processes and structures that limit the rehabilitation potential and may assist in prioritizing investment in systems in which restoration is feasible. Carpenter and colleagues (1999) provided an example of a comprehensive and rigorous scientific approach to classify the restoration potential of eutrophic lakes as irreversible, hysteretic, or reversible. Achieving this level of predictive power in large river ecosystems with multiple anthropogenic pressures is likely to be more complex than in lakes and is a research frontier beyond the scope of this article. However, long time-series monitoring or evaluating historical changes in fish and other aspects of biodiversity can yield powerful insight into the rehabilitation potential of certain ecosystems (see Rinne et al. 2005, Hobbs WO et al. 2012).

Novel, hybrid, and historical ecosystems. The AB concept provides a basis to begin developing new points of reference for the management of novel and hybrid aquatic ecosystems and highlights the conservation value of the remnants of historical ecosystems (table 2). Anthropocene baselines will be most useful as points of reference for evaluating changes in highly modified aquatic ecosystems, such as regulated rivers and wetlands in agricultural or urban landscapes, or where ecosystems are dominated by invasive species.

Novel ecosystems can be distinguished from *historical* and *hybrid* environments by virtue of whether or not changes are considered reversible (Hobbs RJ et al. 2013). The criterion for classifying an AB is that the ecosystem will not be restored to historical ranges of variability (Wiens et al. 2012) because of either socioeconomic constraints on reducing anthropogenic pressures (type I AB) or internal ecological constraints (type II AB; table 2). This is challenging to assess quantitatively (see Carpenter et al. 1999), but ecologists and mangers can make rational judgments about the feasibility of returning aquatic ecosystems to historical ranges of variability using a multiple lines of evidence approach. This judgment will require a synthesis of a broad array of data

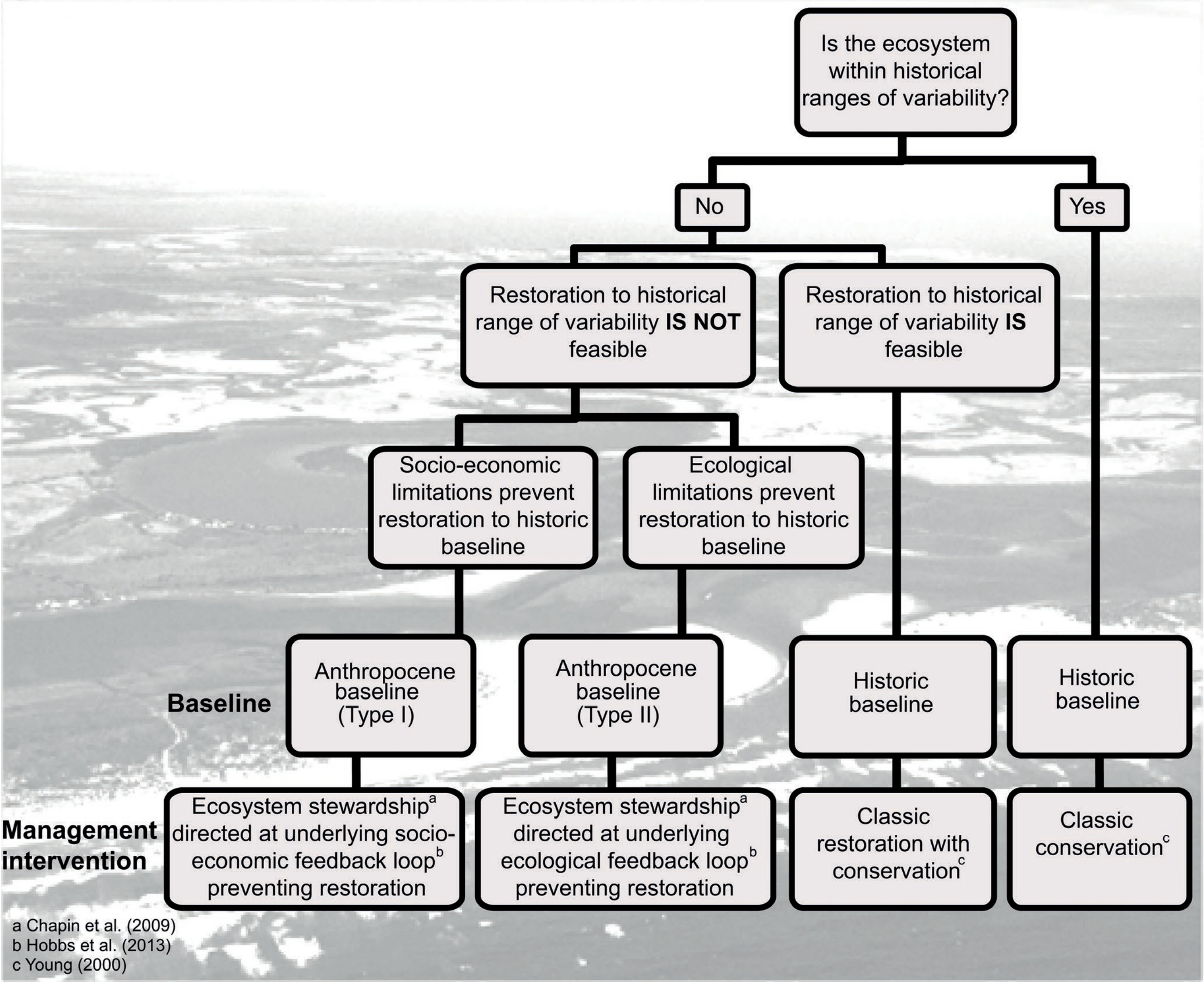


Figure 3. Decision tree for assessing whether Anthropocene baselines (ABs) or historical baselines apply.

sources, such as contemporary environmental monitoring and field data, reference sites, natural archives (e.g., tree rings, calcified structures, sediments, middens, peat lands), historical documents, predictive modeling, long time-series data analysis, and expert opinion (Hughes RM et al. 1986, Jackson et al. 2001, Rinne et al. 2005, Stoddard et al. 2006, Humphries and Winemiller 2009, Dick et al. 2011, Downing et al. 2012, Hobbs WO et al. 2012).

Examples where ABs may apply include the Murray and Darling Rivers in Australia; the Columbia, Hudson, Lower Colorado, Missouri, and Mississippi Rivers in North America; and systems such as the Danube, Yangtze, and Indus Rivers in Europe and Asia. These large rivers and other smaller streams are candidates for consideration as *novel ecosystems* (Hobbs RJ et al. 2013) if (a) they contain abiotic, biotic, and social components which differ from historical conditions and (b) these qualities appear stable over relatively long (e.g., more than 10 years) periods of time. For example, Martis Creek is a highly modified stream in California that is considered to be a novel ecosystem because it has a long history of human-induced alterations to abiotic processes, but it has supported a fish assemblage of five native species and two dominant invasive species consistently for more than 30 years (Moyle 2014). Although similar long time-series data documenting the stability and abundance of invasive species in rivers is relatively rare, the resistance of highly altered aquatic ecosystems to change may be common (Marks et al. 2009, Hobbs WO et al. 2012).

Criticisms of the *novel ecosystem* concept have centerd on the difficulty of objectively distinguishing novel, hybrid, and historical ecosystems and whether changes are irreversible (Murcia et al. 2014). The classification of ABs (figure 3) and associated historical-range-of-variability approaches (Wiens et al. 2012) allows for a more rigorous and quantitative means of distinguishing between novel and historical ecosystems. Other ecosystems may be considered to be in a *hybrid* state (Hobbs RJ et al. 2013), in which classic restoration (*sensu* Young 2000) to within the historical baseline range of variability is considered feasible. Without significant management intervention, these hybrid systems are expected to arrive at AB conditions (table 2).

Distinguishing between hybrid and novel ecosystems will probably be a greater challenge for scientists than determining whether ecosystems are outside of historical ranges of variability. One outstanding question is whether hybrid ecosystems represent a stable state or whether they are an unstable transition between Anthropocene and historical baselines. We hypothesize that hybrid river ecosystems are likely to be among the least disturbed rivers in many areas, including Cooper Creek and the Ovens and Paroo Rivers in Australia; the Fraser, South Fork Salmon, Buffalo, Upper Pitt, and Usumacinta Rivers in North America; and the unregulated tributaries in the Okavango and Congo Basin in Africa. These types of river ecosystems may be candidates for intensive restoration and conservation activities, but in many cases, ABs may still represent the most appropriate point of reference.

Historic baselines will remain useful reference points for the management of remnant ecosystems that are within historical ranges of variability or are expected to return to historic ranges of variability if increases in anthropogenic pressure are limited (figure 3). These types of aquatic ecosystems are unquestionably rare but may include free-flowing river systems and associated aquatic habitats in wilderness landscapes. Examples may include freshwater systems in parts of northern Australia, the Amazon basin, upper middle Mekong, Bhutan, Borneo, New Guinea, and, to a lesser extent, because of climate change, Alaska, Siberia, and northern Canada. In all of these systems, humans are not the obvious dominant ecological force (see Vorosmarty et al. 2010) despite the unavoidable impacts from invasive species and global climate change. These remnant ecosystems should be an urgent priority for global conservation, especially given their high per-unit-area biodiversity value and threat of loss (see box 1, table 1).

Integrating Anthropocene baselines in restoration, conservation, and management. Scientists, managers, and policymakers should critically evaluate whether ecosystems have arrived at an AB (figure 3) and adjust intervention approaches accordingly (table 2). If it is concluded that an ecosystem is at an AB, we suggest that management approaches should shift from traditional conservation and restoration (Young 2000) toward ecosystem stewardship (table 2; Chapin et al. 2009). This approach shifts the focus to social–ecological sustainability rather than returning systems to historical conditions. All management approaches, regardless of whether the point of reference is an AB or a historic baseline, will need to limit human demands (Noss et al. 2013) in order to achieve sustainability. We suggest, however, that the approach to management intervention should vary depending on whether the point of reference is an AB or a historic baseline.

Classic conservation or restoration (Young 2000) approaches may be most effective in ecosystems that are within or near historical ranges of variability (figure 3). Management interventions for ABs will be most effective when underlying social and ecological constraints are addressed. Interventions for type I ABs focus on the social system, whereas interventions for type II ABs focus on the ecosystem constraints (figure 3), but in many cases, both types of ABs will apply. Identifying social and ecological constraints will help guide management approaches to target the underlying mechanisms affecting unwanted changes or maintaining undesirable states. Examples of management intervention for ABs may include creating social incentives (e.g., tax relief for nonirrigated crop production) to limit further increases in human drivers (e.g., water extraction) of ecological change or identifying new social and ecological windows of opportunity (e.g., disturbance events) to maximize desired intervention outcomes.

Management interventions such as environmental flows (Arthington et al. 2006, Poff et al. 2009) and freshwater protected areas will remain central to maintaining ABs. However, the socioecological risks and the potential for negative effects of interventions must be carefully assessed against the risks of taking no action (Hobbs RJ et al. 2011). Unconventional management approaches that are inconsistent with historic conditions (Seastedt et al. 2008) will be required in order to maintain desirable attributes of biodiversity and ecosystem services. For example, in situations in which species are severely threatened, introductions outside of historical distributions or intentional hybridization (and loss of historic genetic integrity) may represent the only means of maintaining wild populations (Jachowski et al. 2015). Given limited financial resources, the AB concept may facilitate site- and taxa-specific prioritization of restoration, conservation, and management activities (table 2). Managers will not waste resources chasing targets that are unfeasible and can work to realize beneficial conditions that are possible under social–ecological circumstances, irrespective of historic precedence.

The effectiveness of management interventions can be assessed using a trajectory approach (Chapin et al. 2009), in which the direction of long-term changes are evaluated against an AB. Distinguishing ABs from historic baselines will be particularly important in degraded systems in which the best reference conditions are in a poor state, thereby falsely suggesting that other degraded sites are in comparatively "good" condition. The reference-condition approach (Stoddard et al. 2006) is currently operational in the United States, and variations are used in river and stream bioassessment around the world. Anthropocene baselines are unlike the "best attainable condition" described by Stoddard and colleagues (2006) because ABs apply exclusively to ecosystems that remain outside of their historical range of variability. We suggest that least disturbed references should be separated into sites where the best attainable condition is within the historic range of variability and those where it is not. Given the high degree of alteration of most managed streams and rivers, we hypothesize that many least disturbed sites will meet the criterion of an AB.

Conclusions

Consideration of ABs forces us to acknowledge modern humanity's domination over most ecosystems. In these environments, we depart from the sense of security grounded by historical references and accept that returning conditions to a past state will not, in the majority of cases, occur. Novel ecosystems (Hobbs RJ et al. 2013) are now common, and the AB concept provides a basis to begin acknowledging new points of reference for managing biodiversity and assessing change in these and other human-dominated systems. Setting clear points of reference for managing human-dominated ecosystems will help alleviate risks that novel ecosystem classification becomes a "license to trash" environments (Murcia et al. 2014). Instead, managers and scientists can work to maintain a new baseline dictated by social–ecological sustainability but will not be constrained to management approaches that fall within the boundaries of historical conditions.

Anthropocene baselines will, in many circumstances, represent a lowered bar—in terms of biodiversity—when compared with preindustrial era baselines. We argue, however, that past and current human activities have already lowered the bar. Many environments simply do not have the same capacity for restoration as they did in the past. In the examples reviewed here (e.g., the mouth of the River Murray, shifts in species' traits, the Aral Sea, lake eutrophication), ABs represent situations where biodiversity and ecosystem services are compromised because of unsustainable natural resource use or the impacts of invasive species. Therefore, we explicitly advocate against setting ABs as targets for management when restoration and conservation toward historical ranges of variability are viable. By classifying ABs, we simultaneously highlight the remnants of historical ecosystems that have escaped modern humanity's domination. These remnants, including free-flowing river systems in wilderness landscapes, are an urgent priority for global conservation.

Anthropocene baselines apply when human (type I AB) or ecological (type II AB) constraints prevent recovery to historical conditions. Resilience (Folk et al. 2004) has been altered, and we suggest that these real-world constraints must be reflected by redefining points of reference for management. Rather than abandoning historical baselines altogether, we argue that it will be essential to compare the biodiversity and ecosystem services that could be generated by a restoration trajectory toward historical reference conditions versus maintaining an AB. Systems, such as those overtaken with invasive species or those with large predators removed, may be resistant and resilient to change, but without historical knowledge of their former potential (see Jackson et al. 2001), the *status quo* and ecosystem services they provide may unwittingly appear more desirable than historic conditions.

The decision to steer biodiversity toward a historic baseline or an AB will ultimately be a socioeconomic choice, not one of science. We suggest, however, that this decision should be guided by empirical evidence concerning the ecosystem services, biodiversity, and socioeconomic trade-offs of maintaining ABs compared with attempting restoration and conservation toward historical baselines. A decision can then be made balancing the sustainability, benefits, and costs of alternative management strategies.

Supplemental material

The supplemental material is available online at *http://bioscience.oxfordjournals.org/lookup/suppl/doi:10.1093/biosci/biv092/-/DC1.*

Acknowledgments

This manuscript benefitted from insight generously provided by Peter Moyle. Many thanks to Simon Watson for thought-provoking, caffeinated discussions and to Saideepa Kumar, David Watson, Robyn Watts, and anonymous reviewers for constructive feedback. This postdoctoral fellowship was funded by the Ecological Responses to Altered Flow Regimes Research Cluster, which represents a collaboration among the Commonwealth Scientific and Industrial Research Organisation (CSIRO) Water for a Healthy Country Flagship, Griffith University, the University of New South Wales, Monash University, Charles Sturt University, La Trobe University, and the Arthur Rylah Institute of the Victorian Department of Sustainability and Environment.

References cited

Adams CC. 1913. Guide to the Study of Animal Ecology. Macmillan.

Allendorf FW, Hard JJ. 2009. Human-induced evolution caused by unnatural selection through harvest of wild animals. Proceedings of the National Academy of Sciences 106: 9987–9994.

Andersen T, Carstensen J, Hernandez-Garcia E, Duarte CM. 2009. Ecological thresholds and regime shifts: Approaches to identification. Trends Ecology and Evolution 24: 49–57.

Arthington AH, Bunn SE, Poff NL, Naiman RJ. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications 16: 1311–18.

Barnosky AD, et al. 2012. Approaching a state shift in Earth's biosphere. Nature 486: 52–58.

Beechie TJ, Sear DA, Olden JD, Pess GR, Buffington JM, Moir H, Roni P, Pollock MM. 2010. Process-based principles for restoring river ecosystems. BioScience 60: 209–222.

Beisner BE, Haydon DT, Cuddington K. 2003. Alternative stable states in ecology. Frontiers in Ecology and the Environment 1: 376–82.

Biggs R, Blenckner T, Folke C., Gordon L, Norstrom A, Nystrom M, Peterson G. 2012. Regime shifts. Pages 609–616 in Hastings A, Gross L, eds. Encyclopedia of Theoretical Ecology. University of California Press.

Burkhead NM. 2012. Extinction rates in North American freshwater fishes, 1900–2010. BioScience 62: 798–808.

Butchart SHM, et al. 2010. Global biodiversity: Indicators of recent declines. Science 328: 1164–1168.

Carpenter SR, Ludwig D, Brock WA. 1999. Management of eutrophication for lakes subject to potentially irreversible change. Ecological Applications 9: 751–771.

Carpenter SR, Stanley EH, Vander Zanden MJ. 2011. State of the world's freshwater ecosystems: Physical, chemical, and biological changes. Annual Reviews in Environmental Research 36: 75–99.

Chapin FS III, et al. 2009. Ecosystem stewardship: Sustainability strategies for a rapidly changing planet. Trends in Ecology and Evolution 25: 241–249.

Courchamp F, Berec L, Gascoigne J. 2008. Allee Effects in Ecology and Conservation. Oxford University Press.

Davies SP, Jackson SK. 2006. The biological condition gradient: A descriptive model for interpreting change in aquatic ecosystems. Ecological Applications 16: 1251–1266.

Dearing JA, Yang X, Dong X, Zhang E, Chen X, Langdon PG, Zhang K, Zhang W, Dawson TP. 2012. Extending the timescale and range of ecosystem services through paleoenvironmental analyses, exemplified in the lower Yangtze basin. Proceedings of the National Academy of Sciences 109: E1111–E1120.

Dent CL, Cumming GS, Carpenter SR. 2002. Multiple states in river and lake ecosystems. Philosophical Transactions of the Royal Society 357: 635–645.

Dick J, Haynes D, Tibby J, Garcia A, Gell P. 2011. A history of aquatic plants in the Coorong, a Ramsar-listed wetland, South Australia. Journal of Paleolimnology 46: 623–635.

Downing AS, Van Nes EH, Janse JH, Witte F, Cornelissen IJ, Scheffer M, Mooij WM. 2012. Collapse and reorganization of a food web of Mwanza Gulf, Lake Victoria. Ecological Applications 22: 229–239.

Folke C, Carpenter S, Walker B, Scheffer M, Elmqvist T, Gunderson L, Holling CS. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annual Reviews in Ecology Evolution and Systematics 35: 557–581.

Heffernan JB. 2008. Wetlands as an alternative stable state in desert streams. Ecology 89: 1261–1271.

Hobbs RJ, Hallett LM, Ehrlich PR, Mooney HA. 2011. Intervention ecology: Applying ecological science in the twenty-first century. BioScience 61: 442–450.

Hobbs RJ, Higgs ES, Hall CM. 2013. Novel Ecosystems: Intervening in the New Ecological World Order. Wiley-Blackwell.

Hobbs WO, et al. 2012. A 200-year perspective on alternative stable state theory and lake management from a biomanipulated shallow lake. Ecological Applications 22: 1483–1496.

Hughes RM, Larsen DP, Omernik JM. 1986. Regional reference sites: A method for assessing stream potentials. Environmental Management 10: 629–635.

Hughes TP, Linares C, Dakos V, van de Leemput IA, van Nes EH. 2013. Living dangerously on borrowed time during slow, unrecognized regime shifts. Trends in Ecology and Evolution 28: 149–155.

Humphries P, Winemiller KO. 2009. Historical impacts on river fauna, shifting baselines, and challenges for restoration. BioScience 59: 673–684.

Ibañez C, Alcaraz C, Caiola N, Rovira A, Trobajo R, Alonso M, Duran C, Jiménez PJ, Munné A, Prat N. 2012. Regime shift from phytoplankton to macrophyte dominance in a large river: Top-down versus bottom-up effects. Science of the Total Environment 416: 314–322.

Jackson JBC, et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–38.

Jackson JB, Sala E, and Alexander KE. 2011. Shifting baselines: The past and the future of ocean fisheries. Island Press.

Jachowski DS, Kesler DC, Steen DA, Walters JR. 2015. Redefining baselines in endangered species recovery. Journal of Wildlife Management 79: 3–9.

Kämpf J, Bell D. 2014. The Murray/Coorong Estuary: Meeting of the waters? Pages 31–47 in Wolanski E, ed. Estuaries of Australia in 2050 and Beyond. Springer.

Lewis SL, Maslin MA. 2015. Defining the Anthropocene. Nature 519: 171–180.

Liittschwager D. 2012. A World in One Cubic Foot: Portraits of Biodiversity. University of Chicago Press.

Marks JC, Haden GA, O'Neill M, Pace C. 2009. Effects of flow restoration and exotic species removal on recovery of native fish: Lessons from a dam decommissioning. Restoration Ecology 18: 934–943.

Micklin P. 2010. The past, present, and future of the Aral Sea. Lake Reservoir: Research and Management 15: 193–213.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-Being: Wetlands and Water Synthesis. World Resources Institute.

Moreno-Mateos D, Power ME, Comin FA, Yockteng R. 2012. Structural and functional loss in restored wetland ecosystems. PLOS Biology 10 (art. e1001247).

Moyle PB. 2014. Novel aquatic ecosystems: The new reality for streams in California and other Mediterranean climate regions. River Research and Applications 30: 1335–1344.

Murcia C, Aronson J, Kattan GH, Moreno-Mateos D, Dixon K, Simberloff D. 2014. A critique of the "novel ecosystem" concept. Trends in Ecology and Evolution 29: 548–553.

Naiman RJ, Alldredge JR, Beauchamp DA, Bisson PA, Congleton J, Henny CJ, Huntly N, Lamberson R, Levings C, Merrill EN. 2012. Developing a broader scientific foundation for river restoration: Columbia River food webs. Proceedings of the National Academy of Sciences 109: 21201–21207.

Neubauer P, Jensen OP, Hutchings JA, Baum JK. 2013. Resilience and recovery of overexploited marine populations. Science 340: 347–349.

Noss RF. 1990. Indicators for monitoring biodiversity: A hierarchical approach. Conservation Biology 4: 355–364.

Noss R[F], Nash R, Paquet P, Soulé M. 2013. Humanity's domination of nature is part of the problem: A response to Kareiva and Marvier. BioScience 63: 241–242.

Palmer MA, Menninger HL, Bernhardt E. 2010. River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? Freshwater Biology 55: 205–222.

Palumbi SR. 2001. Humans as the world's greatest evolutionary force. Science 293: 1786–1790.

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

Pittock J, Finlayson M, Gardner A, McKay C. 2010. Changing character: The Ramsar convention on wetlands and climate change in the Murray-Darling Basin, Australia. Environmental Planning Law Journal 27: 401–425.

Poff LN, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: A paradigm for river conservation and restoration. BioScience 47: 769–784.

Poff LN, et al. 2009. The ecological limits of hydrological alteration (ELOHA): A new framework for developing regional environmental flow standards. Freshwater Biology 55: 147–170.

Reeves JM, Haynes D, García A, Gell PA. 2014. Hydrological change in the Coorong Estuary, Australia, past and present: Evidence from fossil invertebrate and algal assemblages. Estuaries and Coasts. (15 June 2015; *http://link.springer.com/article/10.1007/s12237-014-9920-4*)

Ricciardi A, Rasmussen JT. 1999. Extinction rates of North American freshwater fauna. Conservation Biology 13: 1220–1222.

Rinne JN, Hughes RM, Calamusso R. 2005. Historical Changes in Large River Fish Assemblages of the Americas. American Fisheries Society.

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B. 2001. Catastrophic shifts in ecosystems. Nature 413: 591–596.

Scheffer M, Bascompte J, Brock WA, Brovkin V, Carpenter SR, Dakos V, Held H, van Nes EH, Rietkerk M, Sugihara G. 2009. Early-warning signals for critical transitions. Nature 461: 53–59.

Seastedt TR, Hobbs RJ, Suding KN. 2008. Management of novel ecosystems: Are novel approaches required? Frontiers in Ecology and the Environment 6: 547–553.

Steffen W, Crutzen PJ, McNeill JR. 2007. The Anthropocene: Are humans now overwhelming the great forces of nature? Ambio 36: 614–621.

Stoddard JL, Larsen DP, Hawkins CP, Johnson RK. 2006. Setting expectations for the ecological condition of streams: The concept of reference condition. Ecological Applications 16: 1267–1276.

Strayer DL, Cole JJ, Findlay SEG, Fischer DT, Gephart JA, Malcom HM, Pace ML, Rosi-Marshall EJ. 2014. Decadal-scale change in a large-river ecosystem. BioScience 64: 496–510.

Suding KN. 2011. Toward an Era of Restoration in Ecology: Successes, Failures, and Opportunities Ahead. Annual Reviews in Ecology Evolution and Systematics 42: 465-487.

Suding KN, Gross KL, Houseman G. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology and Evolution 193: 46–53.

Tisseuil C, Cornu J-F, Beauchard O, Brosse S, Darwall W, Holland R, Hugueny B, Tedesco PA, Oberdorff T. 2013. Global diversity patterns and cross-taxa convergence in freshwater systems. Journal of Animal Ecology 82: 365–376.

Tockner K, Bunn SE, Gordon C, Naiman RJ, Quinn GP, Stanford JA. 2008. Flood plains: Critically threatened ecosystems. Pages 45–61 in Polunin N, ed. Aquatic Ecosystems. Cambridge University Press.

United Nations. 2010. Strategic Plan for Biodiversity (2011–2020), including Aichi Targets. United Nations. (4 January 2013; *www.cbd.int/sp*)

Vorosmarty CJ, et al. 2010. Global threats to human water security and river biodiversity. Nature 467: 555–561.

Walters C, Kitchell JF. 2001. Cultivation/depensation effects on juvenile survival and recruitment: Implications for the theory of fishing. Canadian Journal of Fisheries and Aquatic Sciences 58: 39–50.

Wedderburn SD, Barnes TC, Hillyard KA. 2014. Shifts in fish assemblages indicate failed recovery of threatened species following prolonged drought in terminating lakes of the Murray–Darling Basin, Australia. Hydrobiologia 730: 179–190.

Wiens JA, Hayward GD, Stafford HD, Giffen C. 2012. Historical environmental variation in conservation and natural resource management. Wiley-Blackwell.

Young TP. 2000. Restoration ecology and conservation biology. Biological Conservation 92: 73–83.

R. Keller Kopf (rkopf@csu.edu.au), C. Max Finlayson, and Paul Humphries are affiliated with the Institute for Land, Water, and Society, at Charles Sturt University, in New South Wales, Australia. Neil C. Sims is affiliated with CSIRO Land and Water, in Clayton, Victoria, Australia. Sally Haldyz is affiliated with the School of Biological Sciences at Monash University, in Clayton, Victoria, Australia.