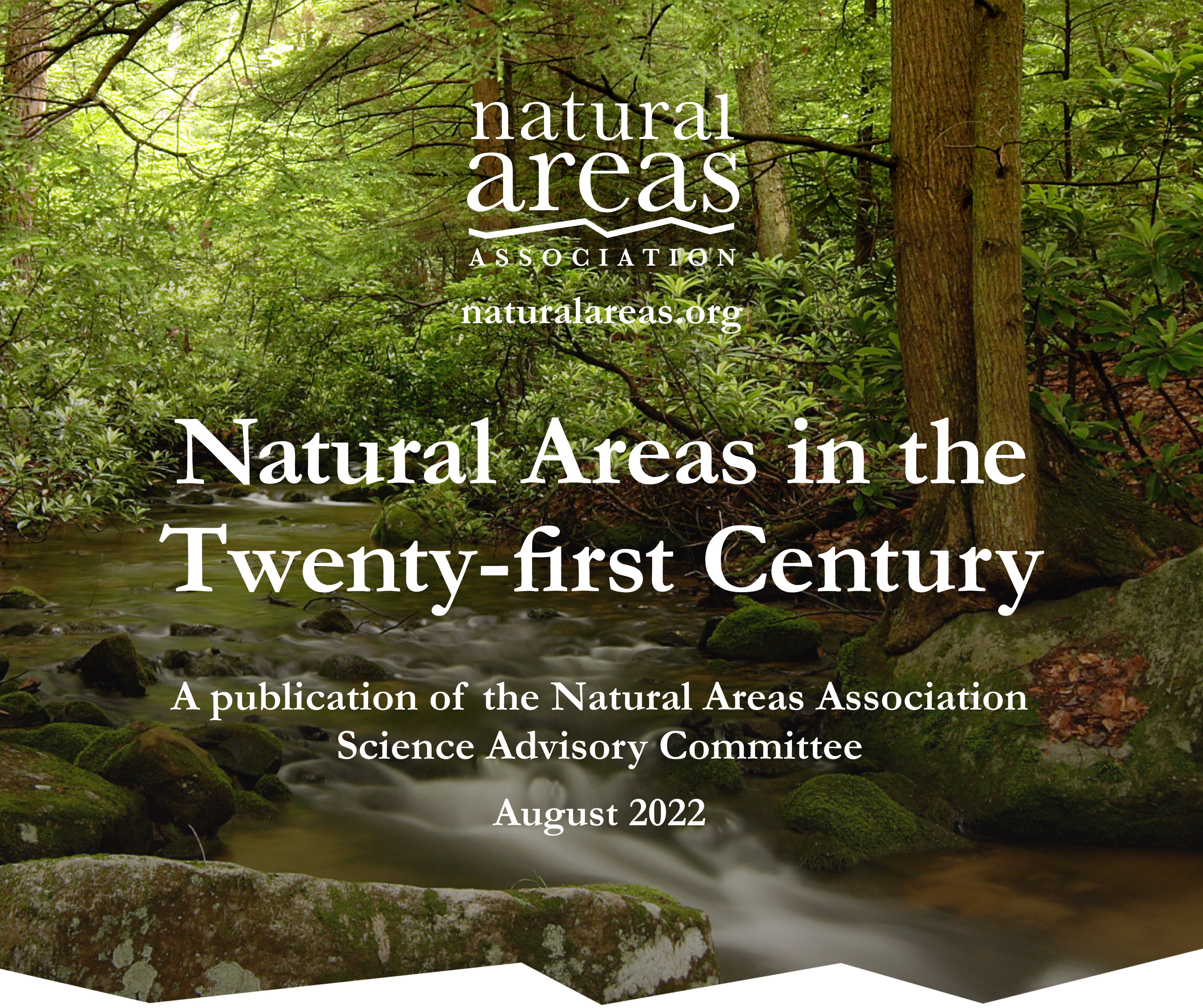

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Natural Areas in the Twenty-first Century

A publication of the Natural Areas Association Science Advisory Committee

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Introduction

The concept of natural areas, as well as the establishment of formal programs for the identification, protection, and stewardship of such areas, is one of several contributions to nature conservation that developed largely in North America, albeit with European antecedents extending back at least to Alexander von Humboldt (Wulf 2015). The world has changed considerably since the natural areas movement began in the early 20th century, as has our understanding of what the future might bring in terms of changed climate and other environmental conditions. As such, it is imperative to reassess the role of natural areas in conservation. Are natural areas still relevant to the public in the 21st century? Do they still serve the purposes for which they were established? Have the values (real and perceived) of natural areas changed over time? How might natural areas be better designed, managed, and marketed to meet changing environmental and social conditions over the remainder of this century?

This paper represents the initial findings of the Scientific Advisory Committee (SAC) of the Natural Areas Association (NAA). The NAA, founded in 1978, is a professional society serving the community of natural areas researchers and practitioners; since 1981, NAA has also published the *Natural Areas Journal*. The SAC was established in 2020 to advise the board of directors of the NAA on issues critical to the organization and its future. Reed Noss was appointed as chair of the SAC. The NAA board intended that the SAC "advises the Natural Areas Association (NAA) Board of Directors and staff on developing critical issues regarding the identification, protection, restoration, study, public uses, and stewardship of natural areas ... [T]he SAC will produce written reports advising the NAA Board of Directors and staff on defined areas of interest to the Board. These written reports developed by the SAC will be used to inform the content and direction of the NAA's four primary program areas: the annual Natural Areas Conference, regional workshops, webinars, and the *Natural Areas Journal*" (SAC Charter unpublished). Members of the SAC were invited by Reed Noss in consultation with the NAA board and staff and were selected based on their expertise in natural areas science and to represent, as much as possible, a diversity of perspectives and geographies.

The NAA board envisioned that the first report of the SAC would "identify gaps between breaking science and conservation practice with regards to the management of natural areas. This report will also contain recommendations for how the NAA could bridge these gaps in order to better equip our members with the most effective tools (e.g., knowledge, technical skills, equipment) currently available to address emerging threats to natural areas." After further discussion and consideration of the results of a poll of natural areas practitioners, the SAC and NAA board agreed that the changing role of natural areas in the landscape and in society, particularly with respect to changes in climate and land use during the 21st century as well as evolving societal norms, is the timeliest topic for this report. We hope it is relevant to the work of natural areas professionals. As discussed later in this report, we recognize that the history of the natural areas movement has been dominated by white people of northern

European ancestry and until recently (with a few prominent exceptions) by males. Making the natural areas profession—and the appreciation of natural areas—more diverse and inclusive is not only ethically correct but may be essential to the survival of natural areas as a public good through this century and beyond.

A Brief History of the Natural Areas Movement

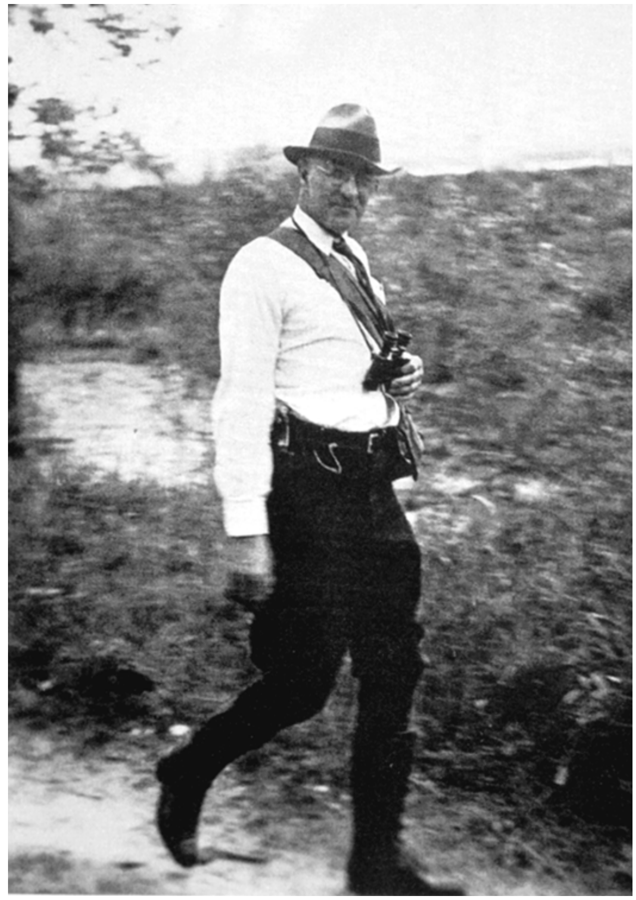

Figure 1. Victor Shelford leading a field trip at Reelfoot Lake, Tennessee, in 1937. Photo by Eugene Odum. From Croker (1991).

To decide where to go, it is worthwhile to understand where we have been. Natural areas conservation on a broad scale in North America began with the Committee on the Preservation of Natural Conditions of the Ecological Society of America (ESA), founded in 1917 and chaired by Victor E. Shelford (Figure 1), who was the founding president of the ESA. "It is a committee on the preservation of nature. Its efforts are directed toward the preservation of natural areas with original flora and fauna (or as nearly so as may obtain) and the maintenance of the natural biotic balance in existing preserves" (Shelford 1926). The overarching charge of this committee was to list all preserved and preservable areas in North America in which natural conditions persisted and to promote their preservation. Shelford's committee drew up maps of the U.S. and Canada and, starting with national parks, identified large areas representative of major ecosystem types. Parks were often proposed for expansion and buffer zones were drawn to surround them. New protected areas were proposed for ecosystems, such as the tallgrass prairie, for which no large parks yet existed (Aldo Leopold Archives, University of Wisconsin-Madison, unpublished). Thus began the natural areas movement (Fell 1983).

The goal of Shelford and colleagues was to preserve a full array of ecosystem types, in as pristine condition as possible, for scientific study. Natural areas were described as "living museums" for research and education. They were recognized by the presence of native vegetation and associated species as well as the relative absence of anthropogenic stressors. As a start, the Preservation committee called for protection of "an undisturbed area in every national park and public forest." This goal quickly expanded into a more visionary resolution to establish “a nature sanctuary with its original wild animals for each biotic formation,” which was proposed by the Preservation Committee and accepted by the ESA Governing Board in 1931 (Croker 1991). This is an early example of the ecosystem representation goal, now a central feature of systematic conservation planning worldwide (Noss and Cooperrider 1994; Margules and Pressey 2000; Groves 2003; Kukkala and Moilanen 2013).

Government agencies in the U.S. quickly became involved in the natural areas movement. Initially the U.S. Forest Service did not differentiate between wilderness, primitive, and natural areas. In 1924, at Aldo Leopold’s urging, the Gila Wilderness in the Gila National Forest of New Mexico became the world’s first designated wilderness area. In 1927 a 4100-acre ponderosa pine forest in Arizona was withdrawn from timber or forage production and became the first natural area—the Santa Catalina Research Natural Area—set aside primarily for scientific study (Moir 1972).

In Canada, where responsibilities for natural resources and public lands lie primarily with the provinces, natural areas protection began with a declaration, “Sanctuaries and the Preservation of Wild Life,” issued by the Federation of Ontario Naturalists and seconded by the Royal Canadian Institute in 1934 (Federation of Ontario Naturalists 1934). This built on the Statement of the Ecological Society of America on Sanctuaries and Reserves, and stated, “in most civilized countries today sanctuaries are being set aside for the preservation of representative samples of the natural conditions characteristic of those countries.” World War II intervened, but in 1942 six organizations convened a conference to discuss Conservation and Post-War Rehabilitation, which reinforced the role of conservation areas as critical elements of watershed conservation planning under Conservation Authorities (Guelph Conference 1942).

A park agency was established in Ontario in 1954, which in 1965 became a participant in the International Biological Programme (IBP), a volunteer effort to document natural areas for possible regulation as ecological reserves (Taschereau 1984). In Ontario the IBP was institutionalized within the province’s parks agency, which conducted systemic ecodistrict and ecoregional studies. This resulted in documentation of “significant natural areas,” of which many were regulated as Provincial Nature Reserves (Zones). More than 500 of those occurring on private lands were extended protections through land-use controls, property-tax relief, and private land stewardship as “Areas of Natural and Scientific Interest.” Simultaneously, regional surveys of Environmentally Sensitive Areas focused efforts on natural areas conservation at local scales (Eagles 1984). Land trusts have focused on securing development rights and stewardship authority on such natural areas. An example is the Nature Conservancy of Canada, whose mission is to protect “areas of natural diversity for their intrinsic value and for the benefit of our children and those after them” (Freedman 2013). As a result, the concept of natural areas was firmly embedded in conservation planning and practice in Ontario and other Canadian provinces.

Protective zoning within Canada’s national parks is not legally required but is a policy that has been confirmed by Parliament. Protective zoning first occurred in 1961 in Point Pelee National Park. Nationally a five-zone system was adopted in 1967. The zones "I. Special Preservation" and "II. Wilderness" together "make the greatest contribution towards the conservation of ecological integrity," by maintaining "a condition that is determined to be characteristic of its natural region" (Parks Canada 2017). On a provincial level, Ontario Parks also first adopted park classes (and zones) in 1967. Its protective zones were Primitive (later Wilderness), Wild River (later Waterway) and Nature Reserve. The goal of Primitive (Wilderness) parks (or zones) was "representative areas of natural landscapes for posterity and … for wilderness recreation activities and for educational and scientific use." This protective zoning recognized "the psychological need, of many people, to know that unspoiled wilderness areas exist" (Killan 1993). A Nature Reserve park (or zone) was required "to represent and protect the distinctive natural habitats and landforms of the province … for educational and research purposes." Despite institutional challenges in delivery, the foundations were well established in Ontario and elsewhere in Canada for appropriately recognizing and stewarding significant natural areas within parks. In Ontario almost all types of natural areas are treated in land-use planning as components of "natural heritage systems" (Riley and Mohr 1994).

The relative vagueness of the term "natural area" was noted early on in North America. As the eminent ecologist Stanley Cain suggested, "I am wholly in agreement with Edward H. Graham (1944), who says that the term 'natural area' is a very useful and realistic one although incapable of exact definition. One virtue of the term is its very indefiniteness. Like the general term 'community,' it does not commit one to the necessity of certain difficult decisions; but it is an even broader term than community, suggesting a recognition of the simultaneous action of all operative factors and the joint existence of such diverse phenomena as organisms and different physical states of the atmosphere, soil, etc. *A natural area, then, is a geographic unit of any order of size with sufficient common characteristics of various sorts to be of some practical usefulness in biogeography"* (Cain 1947, italics in the original).

The Society of American Foresters (SAF) established a Committee on Natural Areas on 5 February 1947, intended "to inventory known natural areas of the nation" and defined natural area as "an area set aside to preserve permanently in unmodified condition a representative unit of the virgin growth of a major forest type primarily for the purposes of science, research, and education. Timber cutting and grazing are prohibited, and general public use discouraged" (Shanklin 1968). The SAF approach was very forest-centric and ignored non-forest ecosystems such as grasslands and shrublands. The importance of natural areas to the forestry profession was stated succinctly by Franklin and Trappe (1968): "Silviculture is based on concepts of plant succession and climax … Natural stands in various successional stages provide a key for development of sound silvicultural practices." According to Moir (1972), "until recently, the (SAF) committee based its evaluation of what natural areas were needed upon concepts of forest cover types, which emphasized dominant timber growth and not necessarily the total assemblage of plants and animals. This conceptual difference between foresters and ecologists often produced difficulties in establishing natural areas."

In May 1966 the U.S. Forest Service Manual provided that the service "will cooperate with other public agencies and professional organizations such as The Nature Conservancy, Society of American Foresters, American Society of Range Management, and Ecological Society of America to establish and maintain an adequate number and variety of research natural areas" (RNAs) (USFS 1966). A joint statement in 1968 by the secretaries of Agriculture and Interior in the Johnson Administration noted that "research natural areas are important as baselines against which man-caused changes can be measured" (cited in Moir 1972). In the same year, the Federal Committee on Research Natural Areas listed 336 RNAs on federal lands in the United States, almost all of them on the national forests (Franklin et al. 1972).

The multi-agency Federal Committee on RNAs persisted through the 1970s and was housed in the Council on Environmental Quality, which broadened perspectives on natural areas. It defined RNAs as follows: "A Research Natural Area consists of a naturally occurring physical or biological unit where natural conditions are maintained insofar as possible" (cited in Franklin et al. 1972). Importantly, the Committee noted that deliberate manipulation, such as prescribed burning and grazing, should be allowed on RNAs and "may be necessary to maintain desired communities or organisms." The committee also noted that RNAs ideally should be "sufficiently large to protect the features of interest from significant unnatural influences" (Franklin et al. 1972). Thus, many key concepts of modern protected area design and management were present in that formulation of RNAs.

Establishment of state natural areas programs in the middle to late 20th century was a consequential development in the natural areas movement in the United States. This began in the Midwest in 1948 where, at the urging of George Fell, the Chief of the Illinois Natural History Survey, Harlow Mills, presented a report to the Illinois State Academy of Science on remnant natural areas. The report stated that "there may be areas in the state, very distinct for some reason, but too small for inclusion in the State Park System as now visualized. These areas may well deserve public ownership and protection in the public interest" (cited in Pearson 2017). Fell quickly provided Mills a brief report describing several such natural areas in Illinois. Fell held a "conviction from the outset that the preservation of remnant natural areas required more than just buying the few odd parcels that might become available; what was required was a comprehensive strategic approach in selection, stewardship, and administration" (Pearson 2017).

Characteristically ahead of his time, in 1948 Fell had written a resolution, which was passed by the Illinois State Academy of Science, to establish a statewide system of nature preserves. The state of Illinois established this system in 1963 and amended it in 1965. The Illinois Nature Preserves System Act defines a natural area as any area retaining "to some degree its primeval character" or has "unusual flora, fauna, geological, or archaeological features of scientific or educational value" and is set aside "for scientific research, education, esthetic enjoyment and providing habitat for plant and animal species and communities and other natural objects" (Moir 1972).

On a more pessimistic note, the governing board of the Ecological Society of America abolished Shelford's Committee on the Preservation of Natural Conditions and his related Committee on the Study of Plant and Animal Communities in 1946 due to concerns about their preservation advocacy. Disappointed but undeterred, Shelford and his colleagues organized an independent group, the Ecologists' Union, to continue the work of the former ESA committees (Croker 1991). A joint report by the Ecologists' Union and the ESA's Committee on the Study of Plant and Animal Communities was published in *The Living Wilderness* (the journal of The Wilderness Society) in the winter of 1950–1951. This report, a sequel to Shelford's *Naturalist's Guide to the Americas* (1926), documented that no protected areas large enough to contain all native animal species in self-maintaining populations existed for deciduous forests, prairies, or lower elevations of the Rocky Mountains in the United States and Canada. Nevertheless, opportunities to create such sanctuaries still remained in some southern swamps, deserts, higher elevations in western mountains, boreal forests, and tundra (Kendeigh et al. 1950–1951). In 1963 the American Association for the Advancement of Science published results of the most comprehensive study of natural areas in the United States to that date (i.e., an update to Shelford [1926] and Kendeigh et al. [1950–1951]). The report advocated an enlarged and better coordinated natural areas program and listed 2400 scientific papers based on research within natural areas (AAAS 1963).

In 1950 the Ecologists' Union was reorganized and renamed The Nature Conservancy. This initially small organization was led by Stanley Cain (president), George B. Fell (vice-president), and Joseph Hickey (secretary-treasurer) (Croker 1991). Beginning with the spirited and uncompromising leadership of George Fell as its unpaid director, The Nature Conservancy (TNC) ultimately became one of the largest and most successful land conservation organizations in the world. Its first stated purpose was to "to preserve or aid in the preservation of all types of wild nature including natural areas, features, objects, flora and fauna and biotic communities" (Pearson 2017). In 1974 Robert E. Jenkins (TNC's Vice-President for Science) developed the basis of the natural heritage methodology and established the first state natural heritage program in South Carolina (Jenkins 1985). The field inventory and database development activities of the state (and in Canada, provincial) natural heritage programs (called conservation data centres [CDCs] in Canada) led to significant advances in the process of identifying and prioritizing natural areas for protection.

Despite its appealing logic, the natural heritage program methodology developed by Jenkins posed a challenge to the prevailing and more informal and opportunistic method of selecting sites (natural areas) for preservation based on their perceived naturalness and scientific values. Sites—typically those that appeared to be undisturbed—were no longer the primary focus of inventory or protection. Rather, the focus was now on "elements of diversity" (specifically "elements of natural biological and ecological diversity"), especially rare species and both rare and representative natural communities. As described by Jenkins (1985), "The Conservancy reversed the virtually universal procedure of inventorying sites for their natural values ... By systematically listing, classifying, and characterizing the elements rather than the natural areas where they occur, the inventories can determine relative endangerment, track down the finest occurrence on the landscape, and identify conservation priorities in the state."

In practice, however, sites were still evaluated by the natural heritage programs. For example, a "survey site" was the location where botanists, zoologists, and ecologists documented what was present. But now potential conservation sites could be identified using knowledge gathered about the location, extent, and condition of the element occurrences (EOs) they contained. Those identified sites might be further prioritized using this information along with knowledge of how rare or endangered the species or communities on the site were thought to be. Up through the 1990s, state offices of The Nature Conservancy commonly prioritized actions for the coming year using an annual "scorecard" of sites in need of conservation action. In the Forest Service, these ecological elements—often referred to as ecological "target elements"—were often represented by SAF or SRM (Society for Range Management) types during regional selection processes (Cheng 2004).

Also in 1974, when the first state natural heritage program was established, natural areas professionals began having annual workshops in the Midwest. At the fourth Midwest Natural Areas Workshop in Indiana in 1977, a proposal to form a Natural Areas Association (NAA) was discussed and a committee was appointed to explore the idea. The following year the committee reported back to the Midwest Natural Areas Workshop in Missouri, where participants voted to create the organization and elected officers and board members to develop bylaws. The bylaws were adopted, and the first full slate of officers and board members was elected at the 6th Midwest Natural Areas Workshop near Minneapolis in October 1979 (Iffrig 1981). The first issue of the *Journal of the Natural Areas Association* was published in January 1981 (Greg Iffrig, editor). By this time, membership in the NAA had expanded outside of the Midwest and included members from northeastern, southern, and western states, as well as Canada. The journal was renamed the *Natural Areas Journal* in 1982. By 1981 more than half of the U.S. states had natural areas programs as well as natural heritage programs (Iffrig 1981). The Natural Areas Association was recognized as the professional society for the staff of natural areas and heritage programs, with membership open "to those involved in the acquisition, preservation, or management of natural areas" (Iffrig 1981). In 1981 John Schwegman was President of the NAA, Richard Thom was Vice-President, and George Fell was Secretary-Treasurer (this was apparently the original slate of officers).

Although we do not have space to discuss the issue in depth here, some tensions soon arose between TNC and some state natural heritage programs. In particular, there was some resistance among natural areas program staff to the natural heritage program methodology, especially its emphasis on inventory and protection of rare species. Some natural areas professionals saw the elements-of-diversity approach as a threat to their conventional site-based evaluations. Schwegman (1981), while he was president of NAA, wrote, “we must not forget that the roots of our movement lie with the science of ecology and the need to protect natural ecosystems which are so important to that field … While I would be the last to deny the value of individual species conservation, I do believe it must be a subordinate part of a natural areas program.”

The last few decades have seen many changes and advancements in the way natural areas in North America are conceptualized, inventoried, designed, and managed. This modern history is too complex to describe in detail here, but it can be gleaned from the pages of the *Natural Areas Journal, Conservation Biology,* and other journals, as well as from such texts as Noss and Cooperrider (1994), Groves (2003), and Groves and Game (2015), and the literature of systematic conservation planning (e.g., Margules and Pressey 2000). Concepts of landscape ecology (Forman and Godron 1981) were incorporated into conservation planning beginning in the 1980s, which spurred increased emphasis on prioritizing conservation sites and strategies across regional landscapes (sometimes within and across ecoregions) as opposed to single sites as the sole focus for conservation assessment and planning (Noss 1983). There was also acknowledgement of broader landscape-level ecological processes that led to increased emphasis on maintaining the “functional mosaics” of natural communities that compose landscapes and ecoregions (Noss 1987a; Poiani et al. 2000).

Connectivity, though at first controversial in conservation planning (Simberloff and Cox 1987), became an important component of conservation plans, in large part due to increased awareness of metapopulation dynamics (Hanski 1998) and the realization that a connected system of natural areas can be a whole greater than the sum of its parts (i.e., by maintaining regional-scale populations or metapopulations that could not persist within any single, isolated natural area or reserve). Ambitious regional networks of reserves, buffer zones, and corridors (e.g., Noss 1987b), which were considered radical and impractical in the 1980s, became well-accepted, at least among conservation scientists, by the late 1990s. In particular, TNC advanced in its planning from large “bioreserves” (largely intact and functional landscapes with compatible human uses) in the mid-1990s (Poiani et al. 2000) to more comprehensive and representation goal-driven ecoregional plans in the late 1990s and 2000s (Groves et al. 2000, 2002; Groves 2003), with the latter usually incorporating regional-scale connectivity. Nevertheless, the science and analytical tools for connectivity planning were still limited during the time (1996–2005) that TNC ecoregional plans were developed.

In the 2020s, landscape conservation plans from TNC and partners are more explicitly considering landscape resilience and connectivity in anticipation of climate change and future land-use trends, drawing from an abundance of ecological literature supporting these approaches in the late 2000s and 2010s (e.g., Hilty et al. 2006; Millar et al. 2007; Heller and Zavaleta 2009; West et al. 2009; Aplet and Cole 2010; Glick et al. 2011; Cross et al. 2012; Groves et al. 2012; Stein et al. 2013). Federal agencies in the U.S. also adopted landscape-level, climate-informed adaptive management frameworks, and new federally funded programs were established during the Obama Administration (Enquist and Jackson 2016), such as US Fish and Wildlife Service–led Landscape Conservation Cooperatives (but see Baldwin et al. 2018), DOI Climate Science Centers, and USDA Climate Hubs. Some large-scale connectivity plans are now incorporated into state legislation with associated funding for land acquisition (e.g., the Florida Wildlife Corridor Act; Main 2021). Moreover, recent national policy seeks to address biodiversity loss and climate change by targeting ambitious goals like conserving 30% of lands and waters by 2030, adopted by the Biden Administration as "America the Beautiful."

What Qualifies as a Natural Area?

"Nature" and "natural" are fraught terms, invoking a number of different and sometimes conflicting meanings (Cole and Yung 2010), which can make it difficult to define "natural area." We favor a broad, relativistic definition of natural area: *"A natural area is an area of land or water of any size where relatively natural geomorphological, ecological, and evolutionary processes predominate over anthropogenic processes and where assemblages of native species in natural communities generally prevail over non-native species."* This definition can encompass the spectrum of common approaches to natural area identification and evaluation, from those centered on natural values and "intact" places, to sites described through systematic description of component natural communities and species. Some natural areas qualify as protected areas or areas with other effective area-based conservation measures (OECMs; CBD 2018), whereas other natural areas are not yet formally protected or conserved. Moreover, valid conservation approaches are not entirely area-based and may focus more on other attributes not necessarily closely tied to area, such as improved stewardship.

Our proposed definition differs marginally from one the NAA board of directors developed in fall 2021 for purposes of strategic planning: "[Natural areas are] terrestrial and aquatic habitats that harbor native rare species or natural communities that are ecologically significant for the protection of biodiversity. The term ecologically significant natural landscapes is used to include those lands and waters that harbor natural area qualities but are not referred to as natural areas by the managing entity." The substantive difference between these two definitions is that the former does not require that a site harbor "native rare species or natural communities" to qualify as a natural area. A representative example of a natural community, or a site with other significant natural features, would qualify as a natural area, regardless of whether it harbors rare species or rare natural communities.

Given this definition, the following kinds of formally designated areas in the United States and Canada may qualify as protected or conserved natural areas, although it is important to remember that not all protected areas are in natural condition and that many natural areas are not formally protected:

- Federal, state, and locally designated Wilderness areas, Wilderness Study Areas, and inventoried and non-inventoried roadless areas
- Research Natural Areas on national forests and other federal lands
- Botanical Areas, Areas of Critical Environmental Concern, Outstanding Natural Areas, NOAA National Estuarine Research Reserves, and other protective designations on US federal lands
- The relatively undeveloped portions of National Parks, National Preserves, National Monuments, National Historic Sites, National Seashores, and other units of the US National Park System and Canadian National Park system
- The relatively unmodified areas of National Wildlife Refuges
- The relatively unmodified portions of UNESCO Biosphere Preserves and World Heritage Sites
- The undeveloped portions of state and provincial parks
- National Wildlife Areas, Nature Reserves, Migratory Bird Sanctuaries, Significant Natural Areas, Areas of Natural and Scientific Interest (ANSIs), and other protective designations in Canada
- State and county preserves, ecological areas, natural areas, and similar designations
- Private preserves, including those of TNC, National Audubon Society, and other nongovernmental conservation organizations
- Conservation easements that meet the “protection of a relatively natural habitat” conservation purpose of the US Tax Code, as opposed to merely the “open space” purpose
- Undeveloped Indigenous (First Nations) lands and various categories of Indigenous reserves
- Key Biodiversity Areas (some of which are protected, but some not)

Because “natural” is a relative concept, for all kinds of natural areas there are two continua: a continuum of naturalness (or quality) and a continuum of protection. A worthwhile objective is to use management and restoration to help guide natural areas toward higher-quality states, and in many cases that will require stricter protection. What constitutes “protection” or “conservation” is a complex and controversial topic outside our scope here. Suffice to say that ideas surrounding these terms continue to evolve rapidly and that commonly applied formal categories of protected and conserved areas (such as IUCN [Phillips 2004] and the US Gap Analysis Program [Scott et al. 2003]) will likely be revised over time.

Role and Function of Natural Areas Historically and Today

It is important to assess to what extent the traditional perceived values of natural areas are still accepted and relevant. Have some formerly cherished values become passé? Have new critical values emerged? Below, we summarize some of the long-recognized values of natural areas and offer some suggestions of emerging values that are likely to become more important within the near future.

Primary Roles and Functions of Natural Areas

Primary roles of natural areas are those that have historically guided the identification, selection, and management of these areas. For natural areas that are formally protected, these primary roles and functions are often stipulated in the enabling legislation.

As Places to Protect Biodiversity

Biodiversity (short for biological diversity) can be defined as "the variety of life and its processes. It includes the variety of living organisms, the genetic differences among them, the communities and ecosystems in which they occur, and the ecological and evolutionary processes that keep them functioning yet ever changing and adapting" (Noss and Cooperrider 1994, modified from Keystone Center 1991). Protection of biodiversity actually was not one of the originally emphasized functions of natural areas, perhaps because the term was not yet in use. Nevertheless, protecting biodiversity was implicit in the work of Victor Shelford's ESA Preservation Committee; its express purpose was "the preservation of natural areas with original flora and fauna (or as nearly so as may obtain) and the maintenance of the natural biotic balance in existing preserves" (Shelford 1926). Biodiversity quickly became a key concept and rallying cry of conservation biology after Thomas Lovejoy introduced the term "biological diversity" in its modern sense in the Foreword of the first textbook on conservation biology (Lovejoy 1980 in Soulé and Wilcox 1980).

The loss of biodiversity, particularly species extinctions, became an issue when biologists began documenting widespread losses. Biologists agree broadly that the current extinction crisis—with species extinction rates estimated as 100 to 1000 times the normal rate (Pimm et al. 2014)—is one of the greatest crises of our time. Extinctions are occurring everywhere including North America. For example, a recent study showed that 51 species and 14 subspecies and varieties of vascular plants have become extinct in the continental United States and Canada since European settlement (Knapp et al. 2021). This is undoubtedly a gross underestimate of the true extinction rate given the dearth of relevant plant surveys, particularly prior to European settlement. Animals also have had high extinction rates, with the rate of loss of vertebrate species over the last century 114 times higher than the natural/background rate (Ceballos et al. 2015). We are clearly well into the Earth's sixth mass extinction event, with losses of vertebrate species since 1980 estimated to be 71 to 297 times greater than at the end of the Cretaceous Period, when the dinosaurs went extinct (McCallum 2015). Importantly, mass extinction is the only modern crisis that is irreversible. Species, once lost, are unlikely to ever be brought back.

Protecting biodiversity is now a well-accepted goal for protected areas and land management in general. Direct destruction as well as fragmentation and degradation of habitat is generally considered the greatest proximate threat to biodiversity, even more so in these times of rapidly changing climate (Noss and Cooperrider 1994; Wilcove et al. 1998; Groom and Vynne 2006; Haddad et al. 2015; Fletcher et al. 2018). Protection, restoration, and management of habitat is therefore the most promising strategy for reducing extinction rates and maintaining the healthy ecosystems and ecosystem services upon which all species, including humans, depend. Protected areas of various types have been the cornerstone of conservation for well over a century (Noss et al. 1999; Watson et al. 2014; Pimm et al. 2018). Protected natural areas serve not only as refugia for species sensitive to human disturbance but also for biodiversity at higher levels of organization, such as natural communities, ecosystems, and landscapes.

Natural areas are not, of course, the sole repositories of biodiversity—hence the increased recognition in recent decades of the need to manage entire landscapes for biodiversity, as we discuss below. By definition, any place in the biosphere has some level of biodiversity, even if just a single population of a single species. The value of natural areas in this respect is that they contain the flora and fauna (and other organisms) characteristic of particular landscapes in a region or, alternately, they contain rare species or other unusual natural features not common in a region. These two values should be seen as complementary.

Ecologists have long been interested in rarity (Preston 1948; Kunin and Gaston 1993). Conservation biologists recognize that, all else being equal, species that are geographically restricted, specialized on uncommon habitats, or present only in small populations are more vulnerable to extinction (Terborgh and Winter 1980; Rabinowitz et al. 1986; Soulé 1987). Local endemic species are especially vulnerable (and valuable) because extinction locally is also extinction globally (Gentry 1986).

For conservation at the population and species level, the following should receive special attention in selection and management of natural areas:

Imperiled and Vulnerable Taxa

Priority should be given to imperiled and vulnerable taxa. Species are ranked for conservation priority by government agencies (under applicable laws such as the US Endangered Species Act and Canada's Species At Risk Act) and by several nongovernmental organizations, including the International Union for Conservation of Nature (IUCN) with its global Red List of Threatened Species, which categorizes species as Critically Endangered, Endangered, or Vulnerable. NatureServe and state/provincial natural heritage programs apply a parallel system at global, national, and state/provincial scales (Stein et al. 2000). The highest-priority species—which generally are assumed to face the greatest threat of extinction—are considered Critically Imperiled. Sometimes species are observed (or projected by models) to be vulnerable to emerging threats such as climate change, although their documented status may not yet reflect this vulnerability. Therefore, additional tools have been developed to assess climate change vulnerability for species. Some species currently thought to be secure may in fact be more vulnerable when we consider the likely impact of climate change. Natural area professionals should monitor such identified species and take appropriate protective actions.

Endemic Taxa and Disjunct and Peripheral Populations

Conservation scientists typically give high weight to narrowly endemic taxa and to hotspots that contain concentrations of endemic taxa (Gentry 1986; Myers et al. 2000; Mittermeier et al. 2011). Species can be endemic at many geographic scales, from small sites to continents. The smaller the scale of endemism, the higher the conservation importance since global extinction could occur with relatively localized disturbance. For conservation purposes, endemism at a scale of site, township, county, ecoregion, or state/province is most meaningful.

Peripheral populations at the edge of their geographic ranges and disjunct populations geographically separated from the remainder of their species' range also are important for conservation. These populations are more likely to be genetically distinct due to selective pressures, such as climate, that differ from those nearer the center of the species' range, as well as from genetic drift (Lesica and Allendorf 1995). Such populations may be of high evolutionary importance; due to genetic differences and reproductive isolation, they may be on a trajectory of becoming new species.

Ephemeral Habitats for Migratory Species

Inventories of natural areas typically focus on resident species—either permanent residents or summer or winter residents. The greatest value of some natural areas is, however, as ephemeral or stopover habitat for migratory insects, fish, birds, bats, ungulates, and other animals. For example, many national wildlife refuges in the U.S. were established to provide wetlands and other habitat for migratory or overwintering waterfowl.

Representative, Underrepresented, or Imperiled Ecosystem Types

Protection, restoration, and management of natural communities or ecosystems constitutes a "coarse filter" approach to conservation, as a complement to the "fine filter" species-by-species approach. It is presumed that the coarse filter will support the habitat conditions and associated ecological processes that sustain the vast majority of species on the landscape over time without requiring a focus on each individual species (Jenkins 1978; Noss and Cooperrider 1994). The fine filter addresses specialized needs of the rare and vulnerable species.

As usually applied, the coarse filter seeks to represent all native ecosystems in a network of conservation lands, with an emphasis on protecting or restoring the high-quality examples of each ecosystem type. Hence, the coarse filter is essentially equivalent to what Shelford was proposing in the early 20th century except for a difference in scale—Shelford's strategy was continental in scope and focused on ecosystems on a biome scale whereas the coarse filter today is typically applied more locally. Representation of all natural ecosystem types (and occasionally those that are nonnative or novel) should be included in natural and protected areas (see following section) for their value to basic and applied science. Natural communities or ecosystems that are currently poorly represented in existing protected areas (Comer et al. 2020) have a high priority for identification and protection. Some may also be "endangered ecosystems" that have declined severely in extent or quality due to human activities (Noss et al. 1995; Comer et al. 2022).

It is important not to overemphasize protection of rare and unique ecosystems, such as glacial relicts or odd edaphic communities, at the expense of regionally characteristic ecosystems and their biota. Often the dominant and characteristic vegetation types have suffered far greater declines than naturally rare communities, which are generally less suitable for agriculture or development. Extent of decline may therefore be at least equally as important as rarity as a criterion for conservation prioritization (Noss 1991). Severe declines of regionally dominant vegetation or foundational species or other strongly interactive species (Soulé et al. 2005) may have more severe ecological repercussions than the loss of very rare natural communities or species. Alteration of species interactions across food webs and disruption of disturbance regimes or nutrient cycles could be consequences (Noss et al. 1995). Hence it is important that all ecosystem types and their component species be represented in natural area programs and not just the rarest ones (Scott et al. 1993; Noss and Cooperrider 1994). That said, due to their high irreplaceability, we do expect rare and unique natural features to continue to have a special place in the natural areas movement.

Although not included within strict definitions of biodiversity, abiotic environmental features determine biodiversity to a great degree. Unusual geological, hydrological, or other physical features such as cliffs, promontories, outcrops (especially of rock types unusual in the region), sinkholes, caves, springs, and steep ravines are often biologically important because of the unique habitat conditions they provide (e.g., Lindenmayer and Franklin 2002). Some contain concentrations of rare and endemic species adapted to such atypical conditions (e.g., Nitzu et al. 2018).

Even if no rare species have yet been documented from such features, their geophysical or geoclimatic values as species habitat are high, especially during climate change. Some karst features, such as sinkholes and other depressions, have been documented as climatic refugia, where species sensitive to warming conditions can persist for long periods of time despite substantial climate change (Bátori et al. 2017). Some unusual substrates (e.g., rocky glades and other edaphic communities) and the unique species assemblages associated with them have microclimates out of equilibrium with the regional climate, therefore potentially having greater stability during climate change (Noss 2013)—see later section on adaptation to climate change. Aside from their biological values, unusual physical features typically have high aesthetic and scenic values and are popular with the public.

Areas of High Ecological Integrity

Natural areas intended to provide examples of ecosystems should possess ecological integrity (Woodley et al. 1993). Ecological integrity has been defined in several ways, but all essentially follow standard dictionary definitions of integrity, e.g., “the state of being whole, entire, or undiminished” or “a sound, unimpaired, or perfect condition” (https://www.dictionary.com). Like biodiversity or naturalness, ecological integrity is a relative concept—it “measures the composition, structure, and function of an ecosystem, as compared with its natural or historical range of variation” (Tierney et al. 2009). In the context of natural areas management, ecological integrity initially received more emphasis in Canada than in the United States; World Wildlife Fund Canada published a report in the mid-1990s with guidelines for maintaining ecological integrity in representative reserve networks (Noss 1995). But subsequent efforts within the public and private sectors in the United States and elsewhere have brought this to the forefront of land and water conservation (e.g., Parrish et al. 2003; Unnasch et al. 2018).

Miller and Rees (2000) observe that ecological integrity is “associated with wild, untrammeled nature and the self-creative capacities of life to organize, regenerate, reproduce, sustain, adapt, develop, and evolve itself.” Despite this seemingly nebulous description, ecological or biotic integrity can be measured. Quantitative indices such as the “index of biotic integrity" (IBI) have been developed and applied with great success to compare aquatic ecosystems in terms of their relative integrity or degradation using "observed" vs. "expected" species composition and abundance measures for fish, macroinvertebrates, and other organisms (Karr and Chu 1999). The "expected" composition and abundance is based on sampling many reference locations to characterize a range of variation in these values that one might encounter.

Terrestrial ecological integrity has proven a bit complicated to measure, although significant progress has been made toward development of various indices either as composite index values (Andreasen et al. 2001) or as component ecological process metrics like natural wildfire regime departure (Barrett et al. 2006), landscape integrity (Walston and Hartmann 2018), or invasive species effects (Bradley et al. 2018). Physical structure or architecture is also a very important measure of integrity in forest ecosystems (Franklin et al. 2018) as well as in other terrestrial and some marine ecosystems. Tierney et al. (2009) interpreted and reported ecological integrity for multiple measures in forest ecosystems of Acadia National Park using "stoplight" symbology: "Good" (green), "Caution" (yellow), or "Significant Concern" (red). This has become a common feature of natural area resource assessments to periodically communicate conditions and support measurable conservation actions.

The concept of ecological integrity has been attacked recently by critics with an anthropocentric or postmodern inclination. Rohwer and Marris (2021) claim that ecosystems are too dynamic to possess integrity and that "any impression of 'wholeness' is an artifact of the brevity of human lives and the shallowness of our historical records." Karr et al. (2022) has rejoined that ecological integrity has been usefully applied in environmental monitoring and assessment for four decades. It arose from a convergence of Aldo Leopold's (1949) widely quoted phrase—"A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise"—and the language of the US Clean Water Act, whose first objective is "to restore and maintain the physical, chemical, and biological integrity of the nation's waters." Karr et al. (2022) provide examples of how the integrity concept has been usefully applied in environmental management, concluding that "multidimensional assessments founded on integrity and calibrated for unique living systems show great past and future promise."

This debate is likely to continue but we would encourage natural areas managers to continue to utilize the concept of ecological integrity because of its demonstrated value. One could say that this debate harkens back to the earlier approaches to natural area definition, with some favoring identification of apparently "natural" and "intact" places, while others favored systematic documentation of species and communities within those places. With the latter approach, repeatable methods for measuring ecological integrity have taken hold, and so it might be no surprise that the value of this concept is better appreciated there.

As Benchmarks or Control Areas for Scientific Comparison with Anthropogenic or More Strongly Manipulated Areas

The value of natural areas as benchmarks—where natural processes dominate—was recognized right from the beginning of the natural areas movement. Shelford's Preservation Committee identified the "living museum" purpose of natural areas for scientific research as the primary reason for establishing a continent-wide network of reserves representing all major ecosystem types. Their goal was having available examples of ecosystem types—minimally influenced by human activities—which would allow scientists to study how nature works when left to itself. As Leopold (1949) commented, "A science of land health needs, first of all, a base datum of normality, a picture of how healthy land maintains itself as an organism … Wilderness, then, assumes unexpected importance as a laboratory for the study of land-health." Later, Moir (1972) observed that "natural areas play a crucial role in the rapidly changing landscape. Most important, perhaps, is that they serve as benchmarks for assessing the extent of man's impact upon diverse land, lake, river, estuary, and coastal environments."

Furthermore, a natural area system that represents a diverse array of ecosystem types allows for comparative research on ecological processes across that entire array. As noted above, to understand and measure ecological integrity, one needs a number of reference locations to characterize a range of variation in native species composition and abundance that one might encounter. Natural areas therefore can serve as key contributors to essential networks of reference sites.

Adaptive management also requires natural benchmarks. Manipulative research in land management benefits from having relatively unmanaged control areas, which represent the same ecosystem types as those being managed, to better gauge the success of management experiments. Natural areas are not ideal controls because no landscape is a perfect replicate of any other, and many human impacts (such as air pollution and climate change) are far-reaching, but they can be the best available and far superior to an absence of unmanipulated areas.

Unfortunately, fire regime departures in many RNAs can undermine their benchmark function. An examination of 64 RNAs on Forest Service lands in California determined that 76% suffered moderate to high fire regime alteration, with most (87%) burning less often than under a pre-settlement fire regime, setting the stage for wildfires to burn at abnormally high severities (Coppoletta et al. 2019). Such findings underscore the need for active restoration and management of many natural areas, rather than simply "letting nature take its course" in reserves too small and isolated—and affected by historical fire exclusion—to manage themselves. It also may point to a mismatch between the spatial scale of these RNAs and the dominant ecological processes at work on those landscapes. In fact, one might require quite large and contiguous RNAs where large-scale natural disturbance is characteristic.

The protected status of natural areas serves long-term research. As noted by Franklin et al. (1972), *"Research Natural Areas are permanently protected by regulation and, therefore, suitable for long-term studies.* On unprotected areas, there is always a risk of disruptions which can destroy many years of work. The value of sites committed to research and protected from outside influences cannot be overestimated, even in short-term research programs." Experimental forests and ranges can also serve this purpose. Unfortunately, research in RNAs is not as common and widespread as it should be, perhaps because many are difficult to access and typically lack facilities.

Some Complementary Roles and Functions of Natural Areas

There are several additional and complementary roles and functions of natural areas relevant to this discussion.

Maintenance of Water Quality

Maintenance of water quality is an important function of many natural areas. A well-vegetated site with a stream running through it, springs or sinkholes on it, or directly adjacent to a lake or stream contributes to regional water quality by reducing runoff of sediments and pollutants. Reduction of storm-water runoff in well-vegetated sites provides benefits to water bodies in the surrounding watershed even when no stream runs through the site or when it is not directly adjacent to a water body.

Historical, Cultural, Scenic, and Recreational Values

Non-biological factors, such as historical, scenic, and recreational values, may be as important as biological values for stakeholders engaged in many conserved natural areas. The key consideration for managers is to ensure that these values are supported in ways that are compatible with the primary or core natural area values present on site. Scenic and recreational values of natural areas are important because people appear to have a psychological need for nature, whether they realize it or not. Just the opportunity to be outside contributes tangibly to emotional well-being. A substantial body of research has confirmed the salubrious effect of nature on human physical and emotional health and intellectual development (e.g., Kellert 2002; Louv 2011; Flies et al. 2017; Oh et al. 2017).

Recreation on natural areas is different from on other types of areas. Although allowable uses vary by managing agency and site, recreation on natural areas is typically nonmotorized, nonconsumptive, and of low intensity. Facilities for visitors are usually minimal or nonexistent, although areas with high visitation rates may have more developed facilities. Some natural areas (or portions of areas) with highly sensitive biodiversity values are closed to the public and can be visited only by permits for scientific research. For most natural areas, recreational access focuses on appreciation of nature and education about natural history. User groups such as native plant societies and Audubon chapters often use natural areas for field trips.

Still other natural areas include essential cultural or spiritual values, such as sacred or cultural sites of Native Americans. Again, conservation of these values is often quite compatible with other natural values, but careful consideration of all perspectives is essential.

Natural Areas as Important Functional Components of Ecosystems and Landscapes

As noted earlier, the science of landscape ecology and its application to natural areas inventory and management increased greatly during the 1980s and 1990s. Historically, most attention from natural areas professionals has been given to species populations and to natural communities defined narrowly (e.g., a calcareous fen) and at a fine spatial scale. Several authors called for more attention to planning on a regional landscape scale (Noss 1983), for an expanded coarse filter that includes functional landscape mosaics (Noss 1987a; Aplet and Keeton 1999; Poiani et al. 2000; Groves 2003), and for generally greater attention to ecosystem dynamics and the landscape matrix (Franklin 1993; Lindenmayer and Franklin 2002) in conservation planning and management. Ecoregion-scale planning of TNC and others in the 1990s–2000s was an acknowledgment that conserving biodiversity would require attention to the entire landscape, and that we needed a much more comprehensive "blueprint" to clarify and prioritize conservation actions at regional as well as local scales.

In Ontario almost all types of natural areas are treated in land-use planning as components of "natural heritage systems," formally defined as comprising "natural heritage features and areas, and linkages intended to provide connectivity and support natural processes which are necessary to maintain biological and geological diversity, natural functions, viable populations of indigenous species, and ecosystems. These systems can include natural heritage features and areas, federal and provincial parks and conservation reserves, lands that have been restored or have the potential to be restored to a natural state, areas that support hydrologic functions, and working landscapes that enable ecological functions to continue" (Ontario Provincial Policy Statements 2020, based on Riley and Mohr 1994).

As noted by Franklin (1993), "Landscape-level issues also need much greater attention. Designing an appropriate system of habitat reserves is one landscape-level concern. Understanding and appropriately manipulating the landscape matrix is at least equal in importance to reserve issues, however, since the matrix itself is important in maintaining diversity, influences the effectiveness of reserves, and controls landscape connectivity." The landscape context of sites, specifically their connectivity or proximity to other protected areas, is a conservation value just as important as the content of sites (Noss and Harris 1986). This consideration has grown more urgent with increased recognition of the need for species to shift their distributions in response to climate change (Heller and Zavaleta 2009)—hence the emphasis in modern climate-aware conservation planning on resilient and connected landscapes (e.g., Belote et al. 2017a).

Maintaining "working landscapes" (e.g., commercial forest lands, rangeland, agricultural areas, pastures) as the matrix in which natural areas are embedded is usually far preferable to having natural areas surrounded by intensive urban or suburban development. In a study in Ontario, the diversity and abundance of birds in forest patches with few or no houses nearby was much higher than in otherwise similar forest patches surrounded by suburbs, probably because the latter had higher rates of predation by house cats, raccoons, and other opportunistic mesopredators (Friesen et al. 1995), similar to findings in southern California (Soulé et al. 1988; Crooks and Soulé 1999). Ideally on public lands, natural areas (e.g., RNAs, wilderness areas) should be surrounded by low-intensity resource production areas, which can serve as buffers (Harris 1984; Noss and Harris 1986), rather than clearcuts or other highly intrusive management activities.

Challenges for Natural Areas in the Twenty-first Century

Natural areas managers now face unprecedented challenges that will continue well into the future. These include land cover/use change and degradation, increasing temperatures, flooding, erosion, drought, nonnative species invasions, and enhanced natural disturbance processes (e.g., insect outbreaks, fire). Many of these issues are not new threats to biodiversity and typically can be managed using conventional conservation approaches (e.g., managing for species viability, removing invasive species, and restoration of fragmented landscapes and altered natural disturbance regimes). Visitor usage rates also can be managed or regulated to mitigate risks to natural and cultural resources. However, when these threats are experienced synergistically, or as extreme events, they can cause increased stress on species and ecosystems, especially those that are already degraded or endangered. Furthermore, the rate of climate and environmental change is accelerating, and many natural area managers are not well prepared to face these challenges.

The Effects of Climate Change and Frameworks for Response

Climate change is already evident in most regions as temperatures and sea levels continue to rise, extreme weather events become more common, and ecosystems are impacted by intensified disturbance regimes. An early discussion of the threats to biodiversity in protected areas posed by climate change (Peters and Darling 1985) notes that “carefully planned and increasingly intensive management” will be needed to minimize species loss and suggests future reserves should be located “where topography and soil types are heterogeneous.” Essentially the same recommendations are being made by conservation scientists today (e.g., Moore and Schindler 2022) but alongside a suite of new recommendations focused on ecological adaptation and resilience.

Over the past two decades, there have been increasing calls for the consideration of climate change in conservation planning and action (e.g., Noss 2001; Millar et al. 2007; Heller and Zavaleta 2009; West et al. 2009; Aplet and Cole 2010; Glick et al. 2011; Cross et al. 2012; Stein et al. 2013; Prober et al. 2019; Peterson St-Laurent et al. 2021). Growing recognition of this problem indicates an urgent need for new skills, tools, and improved understanding of ecological responses and transformations to help make informed decisions for conservation action (Abrahms et al. 2017; Belote et al. 2017a, b; Lam et al. 2020; Hylander et al. 2022). Some researchers suggest that a new, more transformative conservation paradigm is required for the 21st century (Colloff et al. 2017; Prober et al. 2019; Jackson 2021; Peterson St-Laurent et al. 2021; Fougeres et al. 2022). In summarizing this new transformative approach, Moore and Schindler (2022) suggest that “conservation should not just focus on climate change losers but also on proactive management of emerging opportunities.” Focusing on “losers,” of course, is what much of conservation is all about, given our concern about extinctions, but conservation must not stop there. We must try to create an environment where most native species are winners. Importantly, such a landscape will be naturally heterogeneous and well connected.

A review in the Pacific Northwest concluded that natural areas could be ideal for monitoring long-term responses to climate change (Massie et al. 2016); this is due to natural areas having “minimal anthropogenic influences, wide distribution, and proportional representation across several ecological gradients.” An encouraging study from the UK showed that existing protected areas are largely expected to retain their bird species into the future even under an extreme climate change scenario of 4.0 °C warming (Johnston et al. 2013). However, species less mobile than birds are expected to be more vulnerable.

Effects of climate change will vary by region and are described in the 4th National Climate Assessment (NCA4) Volume I (USGCRP 2017) and Volume II (USGCRP 2018), the most recent authoritative assessment of climate-change science, impacts, and risks to the United States (NCA5 is underway and is anticipated to be completed in 2023). Global impacts are most thoroughly reviewed by the Intergovernmental Panel on Climate Change (IPCC). The sixth assessment report of the IPCC is under development, with the draft physical science basis report released in 2021 (IPCC 2021). Given the complexity of the topic, we refer readers to the IPCC and NCA reports rather than attempting to review the vast information on climate change impacts here. Briefly, however, it is well established that most regions will become warmer, but the greatest warming is occurring at higher latitudes, i.e., Canada and Alaska. Rising sea levels will create more frequent inundation of coastal ecosystems and portions of many coastal natural areas will become inundated in coming decades. Warmer ocean waters lead to more evaporation and subsequent increases in atmospheric moisture over many regions and support the development of more intense hurricanes. Extreme rainfall events are expected to continue to increase in frequency in many regions, as are extreme and lengthy droughts. However, the potential changes to future average total precipitation are less certain (USGCRP 2017, 2018). Extreme events are very likely to result in changes in intensity and frequency of disturbance events with consequent disruptions of ecosystems and likely shifts to new stable states.

Significantly for fire-dependent ecosystems and fire managers, the frequency of lightning strikes is predicted to increase over this century. A recent model projects an increase in cloud-to-ground lightning strikes in the United States of 12% per degree Celsius of warming, an approximately 50% increase over the 21st century (Romps et al. 2014). Combined with warmer temperatures, higher rates of evapotranspiration, and drier fuels due to increased drought, more lightning strikes may produce more fires. Nevertheless, fire suppression is currently the primary control on fire regimes in many regions of North America (Mitchell et al. 2014), and intensified suppression efforts can be expected if fire activity increases. This, however, means more work—and more difficult work—for fire managers.

One crucial consideration is that climate change is occurring in landscapes that have been highly fragmented and degraded by human activities. Species that once could have tracked shifting climate zones through natural dispersal no longer can do so. They must now attempt to disperse across landscapes containing fragments of natural or seminatural habitat, and the landscape matrix is occupied by various human land uses that create movement barriers. Also, many invasive nonnative species may fare better than native species under future climate scenarios, although outcomes are uncertain (Hellmann et al. 2008). Invasive exotic species typically have high dispersal capacity, which explains why the ratio of exotic to native grasses in a community is positively associated with the velocity of past climate change (Dukes and Mooney 1999). In general, we can anticipate that responses of invasive species to climate change will be individualistic, as documented for global insect pests (Lehmann et al. 2020).

Perhaps what is most urgent, relative to biodiversity conservation and natural areas management, is that many of Earth's ecosystems are undergoing major transformations with uncertain endpoints. Ecosystem transformations can sometimes be rather abrupt, as when an ecosystem passes some tipping point or is subjected to a major disturbance and flips relatively quickly into an alternative stable state. An example is a fire-excluded pine savanna becoming increasingly less combustible as mesic hardwood trees with nonflammable leaves invade and gain dominance while grasses and other flammable ground cover diminishes.

Eventually a point is reached where the community will not burn, except perhaps a small distance in from the edges or during extreme drought (Noss 2018). Alternately, a woodland may convert to a grassland after invasion by nonnative annual grasses and an increase in fire frequency. Other transformations are more subtle and occur gradually as the ranges of species shift in response to climate change and new sets of species begin to dominate the community (Hobbs et al. 2009, 2014; Jackson 2021). Such changes are generally adaptive and must be accepted by managers, so long as extinction rates of native species are not rising (Moore and Schindler 2022). Increasingly, Indigenous knowledge and local knowledge systems are being recognized and used to further our understanding of ecological transformations (Lam et al. 2020; Long et al. 2020). Sadly, however, the “sense of place” that underpins Indigenous and other place-based conservation practices may be disrupted as the species composition and appearance of places shift with climate change, threatening cultural values and identities (Adger et al. 2013).

Not surprisingly, recent global circulation, or climate, models (GCMs) indicate that expressions of climate stress are highly variable from place to place, and that stress is likely to vary considerably over time during the 21st century. So, while one can presume an overall increase in climate stress regionally, there may be many circumstances—within specific natural areas—where change could be more subtle. This suggests an increasing need to become familiar with forecasted change for ecosystems of interest as they occur locally. This will inform the type and timing of appropriate management response for climate change adaptation.

Various strategies have been proposed for coping with transformations of ecosystems due to climate change. One well accepted framework, called “resist-accept-direct” (RAD), recognizes three basic strategies: resist change, accept change (at some point), or try to direct or guide change in a desirable or tolerable direction (Aplet and McKinley 2017; Jackson 2021; Lynch et al. 2021). Resistance is the most common strategy applied today, as natural areas managers struggle to maintain ecosystems in their historical states, or restore them to those states, even as climate change makes that increasingly difficult. Interventions to resist change can succeed for a while, perhaps for decades—especially if microclimatic refugia are available—and are often appropriate when the existence of globally imperiled communities and species is at stake. Resistance actions seek to avoid loss of native species and ecological integrity, for example by mitigating non-climate stressors (such as habitat fragmentation and associated edge effects) that compound the effects of climate change (Noss 2001). Nevertheless, often resistance eventually becomes futile or at least too expensive to continue over long periods of time, so managers must ultimately switch to another strategy. Thus, identifying the appropriate timeframes for adaptive responses is crucial. An appropriate strategy today may not be useful in 50 years.

The second option, accepting change, has considerable value in many cases. One example is where an essentially unmanaged natural area (i.e., untrammeled wilderness) undergoes substantial alteration with climate change. While perhaps worrisome, those changes are very informative for comparison with natural areas that are actively managed either to resist or direct the impacts of climate change (i.e., toward transformation to a novel ecosystem). A wilderness area, in effect, serves as a control area for management experiments, which as noted earlier is a long-recognized value of natural areas.

The third option, applying management interventions to guide successional pathways and processes toward a novel ecosystem that, while perhaps not as desirable to natural areas managers as the historical state, is at least tolerable because it retains considerable biodiversity and provides ecosystem services. If the manager can ease the transition of the ecosystem to the new state, while retaining as many native species and other natural features as possible, so much the better. As noted by Jackson (2021), “each of these options is fraught with scientific uncertainties and conflicting values,” and “managers may be paralyzed by risks of unintended consequences or failure resulting from intentional intervention.” Such paralysis must be avoided.

Some knowledge of paleoecology helps in making apparent that ecological change is natural and that most modern ecosystems are relatively young, although they may have had similar antecedents during previous periods with similar climate (for example, the mid-Holocene warm period). On the other hand, some ecosystems may have relatively slow rates of change in composition and structure because they occur in relatively stable climatic refugia. Most global hotspots of endemism appear to be regional climatic refugia (Jansson 2003; Harrison and Noss 2017), which is generally good news because they are expected to change less and lose fewer species than regions with higher climate velocity (albeit the absolute loss of species will probably still be higher than during more stable climatic periods).

Various alternatives or nuances to the RAD framework have been proposed. One appealing framework is the “resistance-resilience-transformation” (R-R-T) typology offered by Peterson-St Laurent et al. (2021). These authors acknowledge that all three of these terms are used in confusingly diverse ways. Resilience, in particular, has become a vague concept “with meanings ranging across a spectrum from resisting changes, absorbing changes, and even allowing for transformative changes through self-organization” (Peterson-St Laurent et al. 2021). Still, most conservation scientists likely accept the general definition of resilience as the ability of a system to maintain key functions when disturbed (Gunderson and Holling 2002). Transformation, which is a newer concept basically analogous to directing change, is a more controversial idea because it accepts or even embraces novel ecosystems, a concept that has been critiqued on several grounds, including lack of rigor and clear guidance to practitioners (e.g., Murcia et al. 2014). Nevertheless, in a case study of 104 adaptation projects funded since 2011, Peterson-St Laurent et al. (2021) identified a trend toward acceptance of some form of transformation, although varying across ecosystems (Figure 2).

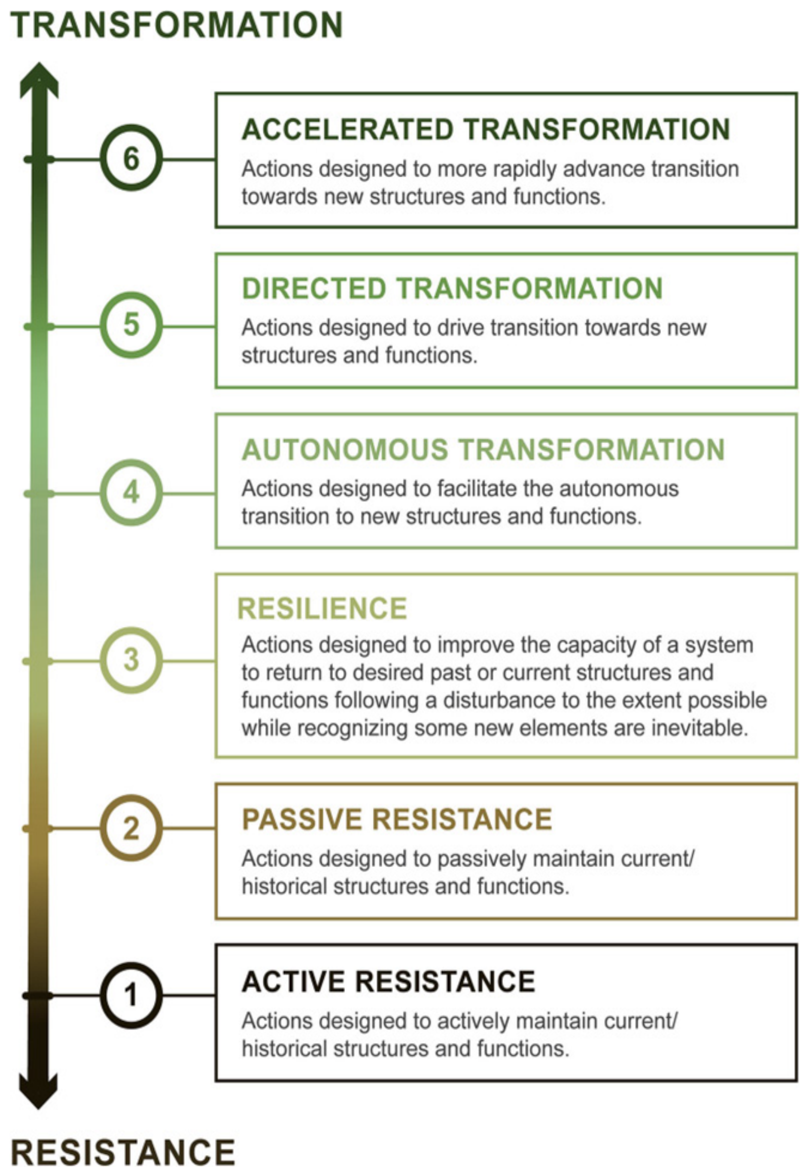

Figure 2. The resistance-resilience-transformation (R-R-T) scale with definitions. The six-point scale ranges from actively resisting changes to accelerating transformation toward new conditions better adapted to an altered climate. From Peterson-St Laurent (2021).

Adaptive Management and What it Means for Natural Areas

Climate change and other environmental change imposes considerable uncertainty on management decisions, making the already difficult job of managing natural areas even more challenging. Over the past few decades, conservation and natural resource managers have increasingly embraced adaptive management as an appropriate philosophy and practice in the context of environmental change. Adaptive management is, in principle, straightforward and difficult to argue against. It is essentially a structured process of learning by doing. In slightly more detail, adaptive management is "an iterative process of gathering new knowledge regarding a system's behavior and monitoring the ecological consequences of management actions to improve management decisions" (Howes et al. 2010). Adaptive management is applicable when resources are responsive to management intervention, but the impacts of those interventions are uncertain (Williams 2011).

Regarding specific management treatments or interventions that managers might apply to a site, for instance to control invasive plants, adaptive management can be summarized by two simple questions: (1) If this intervention was successful, how would we know? and (2) If this intervention was unsuccessful, what would we want to know to do better next time? These questions can be answered through monitoring and adjustment of management approaches based on information obtained from monitoring. However, not every management dilemma or decision is so simple, so adaptive management must be more encompassing than assessing the consequences of specific management treatments. For example, climate change might be causing new nonnative species to invade natural areas in a region. The manager will need to (1) develop a practical forecast of where and in what habitats such invasions are mostly likely to happen, (2) monitor for early detection of the invasive plants, and (3) contemplate options for action when they appear. Above all, it is critical for natural area managers to see themselves and the areas they manage within the context of broader regions, landscapes, and social-ecological systems that are often changing at accelerating rates. Such multi-scale (space and time) thinking and skill sets are increasingly required for the successful management of natural areas in a rapidly changing environment.

Adaptive management is an iterative and cyclic procedure (Figure 3). As shown in the figure, adaptive management involves a cycle of learning and structured decision-making. The consequences of management decisions are evaluated through monitoring, which promotes further learning and improved management decisions.

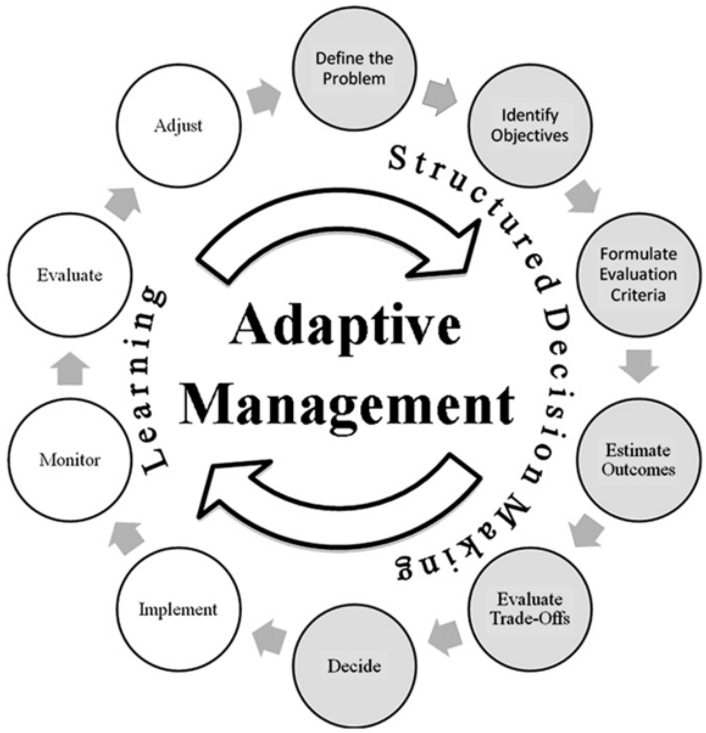

Figure 3. Adaptive management is learning by doing in a formal iterative process that acknowledges uncertainty and achieves management objectives by increasing knowledge of a system through a structured feedback process. Integral to adaptive management is both a learning process and a structured decision-making process. From Allen et al. (2011).

In initiating an adaptive management program, natural areas managers must begin by identifying the most urgent problems requiring management attention. Among the major issues (discussed in the following section) some are more urgent than others and their relative level of urgency will vary among sites and over time. The next step is to identify the management activities that are good candidates to be successful based on prior experience and the best available science. It is also useful to identify alternative treatments in case the selected treatment is not sufficiently successful. An example of a promising framework for decision-making is the National Park Service's Natural Resource Condition Assessments, which are followed by their Stewardship Strategies. The condition assessments document current conditions, evaluate trends, identify critical data gaps, and help managers understand how drivers and stressors influence conditions (https://www.nps.gov/orgs/1439/nrca.htm). The assessment results feed into prioritizing strategies and actions in the park; both are iterative.

Some form of ecological monitoring program should be in place prior to initiating any treatment. This might be as simple as setting up GPS-linked photo stations for a series of "before and after" images, or it might require more quantitative sampling, such as measurements of cover, height, and distribution of various plant species of conservation concern or surveys of birds, herpetofauna, or butterflies. Once monitoring stations have been established, the initial treatments can be implemented.

The next step is to monitor and evaluate the effects of treatments. The evaluation can be a challenging step, as it sometimes can be difficult to assess the significance of the observed responses, such as changes in soil properties. Proposed changes in treatments then can be designed, implemented, and repeated in the adaptive management cycle. A key need is an institutional structure that can and will sustain the monitoring program, analyze the data gathered, and report the results so that managers and others inside and outside the organization can learn from the process.

Given the challenges of adaptive management, managers might be tempted to throw up their hands in confusion and just return to business-as-usual management (i.e., without ecological monitoring and evaluation). But again, adaptive management is inherently a simple and flexible process. If experimental design, quantitative analysis, and ecological modeling are beyond the means of the manager, more basic approaches to learning by doing are often just as acceptable. As observed by Allen et al. (2011):

> Ironically, the confusion over the term "adaptive management" may stem from the flexibility inherent in the approach, which has resulted in multiple interpretations of "adaptive management" that fall along a continuum of complexity and a priori design. Adaptive management is not a panacea for the navigation of "wicked problems" as it does not produce easy answers, and is only appropriate in a subset of natural resource management problems where both uncertainty and controllability are high. Nonetheless, the conceptual underpinnings of adaptive management are simple; there will always be inherent uncertainty and unpredictability in the dynamics and behavior of complex social-ecological systems, but management decisions must still be made, and whenever possible, we should incorporate learning into management.

Guidance for Responding to Climate Change in Natural Areas Management

Over the past decade, there have been numerous calls to not only consider climate change in the conservation planning process, but also to actively invest in the implementation of climate adaptation actions. Here we summarize many of these adaptation approaches in the context of 21st century natural areas conservation.

Given the conundrum of options, none of which is entirely satisfying, some **best management practices** (or at least guidance) for addressing climate-driven environmental change include the following:

- Rather than only considering climate exposure in assessments of vulnerability to climate change, as if often done, also consider the two other components of vulnerability: sensitivity and adaptive capacity (Butt et al. 2016). Adaptation should be understood in a broad sense that includes evolutionary, ecological, and social changes that are likely to reduce the vulnerability of ecosystems to climatic disruption (Moore and Schindler 2022).

- Recognize that climate change is not just a long-term, gradual threat; rather, changes in the frequency and magnitude of climatic extremes are an immediate threat (Butt et al. 2016) and major disturbances linked to climatic change may result in drastic near-term change.
- For unique and highly irreplaceable natural communities and species, such as those that are narrowly endemic or ranked as imperiled at global or state scales, resistance is the preferred option for as long as it can be maintained without unreasonable effort or expense (Millar et al. 2007; Millar and Stephenson 2015).
- Wherever possible, identify and protect climate refugia, which range in spatial extent from small, localized habitats such as sinkholes, seepage areas, north-facing slopes, and edaphic communities (hypothetically) to entire landscapes with relatively stable climates due to topographic heterogeneity, proximity to moderating ocean currents, disturbance regimes (such as frequent fire) that produce resilient ecosystems, and other factors (Noss 2001; Dobrowski 2011; Keppel et al. 2012; Bátori et al. 2017; Harrison and Noss 2017). Recognize, however, that an overly prescriptive approach to identification and prioritization of refugia may compromise maintenance of heterogeneity across landscapes (Moore and Schindler 2022). Ecological heterogeneity is key to adaptability, and maintaining the processes that generate heterogeneity is critical.
- Give greater consideration to the roles species play in the functioning of your targeted ecosystems (Nock et al. 2016). Which are the nitrogen fixers? Who are the pollinators? What species provide the physical structure or some "keystone" function? While one might anticipate that species may be lost over time, maintaining or restoring diversity among functionally significant species groups will reduce your risk.
- If or when resistance becomes futile, managers should still collectively try to prevent extinctions of any species, in part because the future roles of species in ecosystems might be different from what they are today. As Aldo Leopold put it: "To keep every cog and wheel is the first precaution of intelligent tinkering" (Leopold 1949). This will require coordinated actions among managers in different locales, for example to facilitate range shifts in response to climate change. It may also require keeping species in captivity (zoos and botanical gardens), at least temporarily, as well as assisted migration/colonization, where populations are translocated to areas that are becoming more suitable than their current ranges due to climate change (Brodie et al. 2021). However, assisted migration should only be attempted after a thorough assessment is made of the risks, feasibility, and likelihood of success, as well as potential impacts to the recipient ecosystems (Richardson et al. 2009; Schwartz et al. 2012).
- With species dispersing to newly suitable habitats, the definition of what is native to a region must be re-evaluated. As pointed out by Moore and Schindler (2022), "species movement into new habitats has always been key to the biosphere's adaptive response to a changing world, and protectionist perspectives could hinder community adapta-

tion. Perhaps if species are following their projected climate trajectory, then they should be considered 'proactively native.'"

- For directed-change interventions, the most defensible endpoints are usually those that are predicted by the best available climate science for the region in question. Encouraging development of ecosystems that are compatible with projected climatic and disturbance regimes is a wise strategy. It is important, however, that managers understand that such projections are hypotheses, and such interventions are best carried out in an experimental fashion (see below), within a bet-hedging philosophical framework (Safford et al. 2012).
- As recommended by Jackson (2021), an experimental approach is desirable, where managers create "broadly framed, experimental adaptive management portfolios," which include combinations of resistance, acceptance (intervention-free control areas), and a suite of directed-change interventions. Based on the experiments, those interventions (or lack thereof) that lead to unacceptable results are abandoned, and those that produce acceptable biodiversity outcomes are continued. Coordination among managers from different agencies will be critical, as not all types of interventions can be done in any single management jurisdiction.
- Rather than focusing solely on the direct impacts of climate change, recognize that it interacts with other threats such as land-use change (hence the need for more protected area and stronger zoning ordinances), invasive species, and changing disturbance regimes (Butt et al. 2016).
- Avoid simplistic "solutions" to climate change, such as massive tree-planting for carbon sequestration. Afforestation of natural and seminatural grasslands is a major threat to global biodiversity (Veldman et al. 2015, 2019). Natural areas professionals should be outspoken opponents of such popular but naïve programs.
- In general, terrestrial ecosystems sequester and store as much as one-third of CO_2 emissions arising from anthropogenic activity, and land managers should increasingly consider these roles in their planning and management (Canadell et al. 2007), for example by encouraging the restoration of ecosystem types that store substantial carbon (which include not only forests, but grasslands, peatlands, and mangroves, among others; e.g., Veldman et al. 2015).
- Geophysically or geoclimatically diverse landscapes, with heterogeneous topographic and edaphic conditions, offer opportunities for species to adjust to climate change by moving relatively short distances into newly favorable habitats (Ackerly et al. 2010; Anderson and Ferree 2010; Beier and Brost 2010; Anderson et al. 2015). Therefore, establishment of new protected areas should seek locations in such landscapes (Albano 2015). Existing federal protected areas in the conterminous U.S. do not reflect climatic diversity well, with the most common climate types particularly underrepresented (Batllori et al. 2014).
- Conservation strategies under climate change should be diverse and based on a simultaneous evaluation of conservation values and climate vulnerability (e.g., climate

velocity). For example, areas with high conservation value and low climate vulnerability may offer the most promising opportunities for new protected areas, whereas areas with high conservation value and high vulnerability are more challenging to manage and require maintenance of connectivity and protection from additional stressors such as intensive development or resource extraction (Belote et al. 2017b).

Invasive Nonnative Species Control

Most natural areas suffer to some degree from invasions by nonnative plant species and sometimes animal species. The level of invasion and dominance by nonnatives varies tremendously among sites, however, as well as among different portions of individual natural areas. Hundreds of studies of plant invasions and their impacts on native communities have been conducted, but the vast majority have been limited in duration to a few years at most. Thus, we have a poor understanding of how interactions between native species and nonnative species may change over time (D'Antonio and Flory 2017). Invasive animal species also can be significant problems, particularly when they represent an aggressive top predator that native prey species have never had to deal with before.

Managers of natural areas often assume that all nonnative species are bad and should be eliminated as soon as possible. This is not uniformly sound policy. Evidence suggests that "many of the claims driving people's perception that introduced species pose an apocalyptic threat to biodiversity are not backed by data" (Davis et al. 2011). Indeed, many studies have found that some nonnative species play useful roles in ecosystems, often substituting for native species that have experienced population losses or gone extinct, and can actually increase native biodiversity (Davis et al. 2011). Moreover, management to eliminate invasives and restore native plants can have unintended negative consequences on rare native species of conservation concern (Buckley and Han 2014; Casazza et al. 2016). Also, as noted above, species from warmer regions that disperse naturally into formerly cooler regions should not be considered nonnative; species ranges are inherently dynamic over time and climate-driven dispersal is adaptive.

On the other hand, abundant evidence suggests that nonnative species often can have devastating impacts on native biodiversity, albeit the evidence base on which to make management decisions is often limited (Hulme et al. 2014). One of the most problematic impacts stems from the effects of nonnative plants on disturbance regimes, which in turn affect the structure, composition, and function of the ecosystem in multiple ways. Exotic annual grasses not only are highly competitive with native vegetation (Humphrey and Schupp 2004), they also are often highly flammable and increase the amount and continuity of fine fuels as well as the length of time that these fuels are dry enough to burn (Knapp 1995; Davies and Nafus 2013). A tragic example of this phenomenon is the invasion of sagebrush steppe in the Intermountain West by cheatgrass (*Bromus tectorum*; Bradley et al. 2018).

Some recent studies, however, offer a hint of optimism for natural areas managers. One of the few long-term studies (8 years) of invasive impacts showed that the nonnative Japanese stiltgrass (*Microstegium vimineum*) dominated the biomass of research plots in a natural area near Bloomington, Indiana, for the first 4 years, but then declined to just 2% of the biomass after 8 years in plots both with and without fire (Flory et al. 2017). Another study, in northern Nevada, showed that invasive cheatgrass initially has strong negative impacts on native perennial grasses, but it exerts strong selective pressures on these plants, such that after a few years the native grasses had evolved traits that allowed them to persist in cheatgrass-invaded areas (Leger and Goergen 2017). Therefore, what seems to be an extreme invasive problem at one point in time might turn out to be less dire than originally thought.

Clearly there is a need for more research and monitoring of invasive species to inform adaptive management interventions. Based on existing evidence, the following are some **best management practices** for invasive nonnative species on natural areas:

- Not all nonnative species are invasive, and not all nonnative or invasive species have significant detrimental impacts on native ecosystems. Therefore, it is critical to gather evidence through research and monitoring to determine which nonnative species should be eradicated or controlled and which can potentially be left in place. This is a cost-effective strategy, as controlling invasives can be expensive. Sometimes we simply have to learn to live with invasive exotics (Davies et al. 2021).
- In some cases, however, the impacts of an invasive nonnative species are so obvious (though perhaps only from studies or experience elsewhere) that the most prudent management action is to try to eliminate such species as quickly as possible. Nevertheless, monitoring of untreated areas should be pursued, as it may turn out that the impacts of some invasives decline over time as native species evolve traits to escape their impacts or outcompete them.
- Remember that native species can be invasive as well, for example oaks and other hardwoods invading fire-excluded pine savannas (e.g., Brockway and Outcalt 2000).
- Be careful that restoration treatments to remove exotics and restore native plant cover do not harm native species of conservation concern.
- Note that climate change may work for or against species invasion. Particular invasives may be favored or disfavored by emerging conditions that are warmer but wetter vs. drier, so keeping current with climate change trends in your area can provide important insights for management.
- It is important to monitor the effects of invasive species management to determine if expected responses of native species to management actually occur. Studies show that native plant species do not necessarily recover following control of an invasive nonnative species, and in some cases the treated nonnative species is replaced by other nonnatives (Reid et al. 2009). These findings suggest the need for other actions to increase recovery potential for native species after control of the invasive.

- The optimal strategy for addressing nonnative plant invasions may be to develop and maintain a natural community with high ecological integrity and resistance to invasion (Sheley and Krueger-Mangold 2003).

Viability of Species of Conservation Concern

Maintaining viable populations of native species (as many as possible) is a fundamental goal of natural areas management (Noss and Cooperrider 1994). Because field biological surveys of most preserves and other natural areas are incomplete, undoubtedly more species of concern will be found in any given natural area after additional surveys (which are virtually always highly desirable). Many species of conservation concern will require species-specific management and recovery actions, but the following **best management practices** have considerable generality:

- Strive to maintain ecologically effective populations of species of conservation concern, not just minimally viable populations. Species exist in communities and ecosystems and their interactions with other species and processes will vary with their abundance.
- Species of conservation concern generally need to be prioritized for management attention. Important criteria for prioritization (which should be considered in combination) include (1) endemism—those species endemic to the smallest geographical area have highest priority; (2) extinction risk—those species at greatest risk of extinction in the near future have highest priority; and (3) ecological role—those species that are strongly interactive (e.g., keystone species, ecological engineers; Soulé et al. 2003, 2005) are most important to maintain in ecologically effective populations.
- The consideration of conservation needs and necessary actions for large groups of species can be simplified by clustering species according to shared ecosystem types or geophysical habitats, shared threats, or shared functional traits (Clark and Harvey 2002; Kooyman and Rossetto 2008; Noss et al. 2021).
- There is no substitute for intensive field monitoring of populations of conservation concern by highly qualified field biologists, using the most appropriate survey techniques for the species in question. This said, modern techniques such as satellite imagery and environmental DNA (eDNA) are demonstrating their value in augmenting field surveys.

Landscape Context (e.g., Connectivity, Matrix Effects)

Natural areas and preserves do not exist in isolation, although they may be described this way in fragmented landscapes. Natural areas vary in their landscape context, in their types of interactions with surrounding lands, and in their level of connectivity to other conservation lands. As noted earlier, landscapes need more attention in conservation biology and natural areas design and management (Franklin 1993). Natural areas managers have some potential options (pending funding) to improve the landscape context and connectivity of its preserves. **Best management practices** include the following:

- Familiarize yourself with any regional place-based conservation plans in your area to better understand the relative significance of your area and of nearby areas that may be targets for conservation actions.
- Work with landowners surrounding a preserve to encourage land use and land management practices that are compatible with the conservation mission of the preserve.
- Work with land trusts to encourage acquisition of conservation easements on lands surrounding preserves and linking them to other conservation areas.
- Work with federal, state, and local agencies to encourage them to acquire lands (fee simple or easements) to improve the landscape context and connectivity of preserves.
- It might appear that natural areas managers have little control over changes in the landscape context around their preserves. While this is sometimes true, natural areas professionals at higher administrative levels may have influence with county planning and zoning departments and with land trusts that purchase (or receive through donations) conservation easements.

Fire and Other Disturbance Management

In addition to climate and substrate (geology and soils), fire and water are the two ecological factors that have the greatest influence on the distribution, structure, and composition of upland and wetland ecosystems across much of North America. Fire is perhaps best viewed as an ecological driver that shapes ecosystems and typically enriches biodiversity at one or more spatial extents. From this standpoint fire is intrinsic or endogenous to the ecosystem and is promoted by species in the community through coevolved vegetation–fire feedbacks. These feedbacks depend on functional traits of species, such as high flammability of live or dead leaves (Mutch 1970; Beckage et al. 2009; Fill et al. 2015). Other disturbance factors such as wind, landslides, floods, and the activities of megafauna (e.g., trampling) are also important in many communities, although we do not have space to discuss each of these factors here.

It is critical to recognize that plant and animal species are not adapted to fire *per se*, but rather to the particular fire regimes with which they evolved (Keeley et al. 2011). If a fire regime changes sufficiently from a species' evolutionary experience, population decline and potential extinction become likely. A century or more of fire exclusion—in combination with other factors including climatic change—has altered many North American ecosystems. In many cases, regime shifts have occurred, for example as pine savannas shift to the alternative stable state of hardwood forest (Harper 1911; Noss 2018). In much of western North America, the legacy of fire exclusion and other land uses has resulted in forests (especially lower-elevation dry forests, such as ponderosa pine [*Pinus ponderosa*]) that are much denser than prior to settlement and therefore more likely to experience severe stand-replacing wildfires, mortality during drought, and bark beetle outbreaks (Covington and Moore 1994; Schoennagel et al. 2004; Noss et al. 2006; Hagmann et al. 2021).

The first principle of fire management is to establish an understanding of the natural wildfire regime in your area. Maps and models from the LANDFIRE program can assist with this (https://www.landfirereview.org/search.php). Texts documenting historical and current (and sometimes projected future) fire regimes are also available for many parts of North America (e.g., Agee 1993; Safford and Stevens 2017; van Wagtendonk et al. 2018; Greenberg and Collins 2021). Such information is fundamental to developing prescribed fire or wildland fire use (managed wildfire) strategies that best mimic the fire regimes—whether driven by lightning or human ignitions—that shaped the evolution of species and the assembly of communities across the target region. improvements recommended for fire management include introducing a degree of pyrodiversity in all components of the fire regime and avoiding intensive soil-disturbing actions, such as plowing fire lines (fuel breaks) too deep, too wide, or in too high density, which increases nonnative species invasions, among other problems (Noss 2018). **Best management practices** for fire include the following:

- No single correct answer exists to any question about fire management. It all depends on the ecological and practical context. Nevertheless, some practices, techniques, and burning regimes have proved more successful in meeting ecological goals than others. Take an evidence-based approach to decisions about fire management.
- Best management practices depend in part on the specific objectives of prescribed burning. Common objectives include (1) resource management for forestry, range, and wildlife production; (2) reduction of hazardous fuel loads to lower the probability of occurrence or severity of undesired wildfire; (3) creation of suitable habitat for threatened and endangered species; and (4) restoration or maintenance of native ecosystem structure, function, and composition. These objectives are not mutually exclusive, but the emphasis given to each varies widely.
- Set fire-return intervals specific to natural communities. Some natural communities, such as dry prairies and some pine savannas in Florida (Noss 2018), burned from lightning ignitions as often as every 1–2 years, whereas other fire-prone or fire-dependent ecosystems had mean fire-return intervals on the order of decades or longer.
- Restoration burns in communities that have lacked fire for long periods and have developed dense understory or midstory vegetation must often be preceded by active treatment of fuels, such as cutting (thinning), chopping, mulching, or mowing. Especially for ecosystem types naturally characterized by low-severity surface fires, active treatments bring fuels into the herbaceous layer, where they can then be consumed by fire, promoting recovery of suppressed groundcover (Menges and Gordon 2010). Mechanical treatments have ecological costs, however, such as soil disturbance and compaction and increases in nonnative plants and animals (e.g., fire ants [*Solenopsis invicta*]).

- Chemical treatments (herbicides) are often needed to supplement mechanical treatments. However, mechanical and chemical treatments are typically 10 to 20 times more expensive than controlled burning (Waldrop and Goodrick 2012). No mechanical or chemical treatment can serve as a substitute or true surrogate for fire. They are best considered pretreatments for fire and should be applied cautiously and minimally.
- In communities that have suffered long periods of fire exclusion, the midstory (and sometimes canopy) that developed is sometimes dense and tall enough that revenue can be generated by harvesting and selling the timber, while also preparing the site for safe prescribed burning. An accurate justification for this activity is not only that it facilitates burning, but that the invading trees, while native, are also invasive in this context and have created stand densities much higher than normal.
- Pursue pyrodiversity, but cautiously. Pyrodiversity can be defined as variation in fire regimes in time and space. It has been recommended as a bet-hedging strategy that promotes coexistence of species with disparate life histories and requirements with respect to fire (Menges 2007). The assumption that "pyrodiversity begets biodiversity" (Martin and Sapsis 1992) must be examined critically, however. Responses of biodiversity to increases in one measure of pyrodiversity could differ substantially from responses to other measures, and it is doubtful that maximum pyrodiversity by any measure is a legitimate management goal (Noss 2018).
- A precautionary and evolution-informed approach to pyrodiversity would apply variability in fire frequency, seasonality, and other components of the fire regime, but burn most often at the frequency, seasonality, and intensity that was most common under a natural lightning-fire regime or long-term indigenous fire regime.
- Burning on consistent intervals is probably not optimal for maintaining biodiversity. Fire-return intervals substantially longer than the mean or median interval occurred occasionally under a lightning-fire regime, and these windows of time without fire provide temporal refugia to species sensitive to very high fire frequency.
- Leave fire refugia, but not too many. Controlled burns are usually less patchy than lightning fires and leave more homogeneous postburn conditions (Ryan et al. 2013; Noss 2018). Spatial heterogeneity in postfire vegetation contributes to biodiversity, in large part because patches of unburned or lightly burned vegetation serve as critical refugia or microrefugia to plant and animal species relatively sensitive to fire. Maximum patchiness, however, is not an appropriate management goal. The goal is to create a degree of heterogeneity in vegetation that simulates the pattern produced by a natural fire regime.
- When declining populations of highly imperiled and fire-sensitive insects are the conservation focus, permanent non-fire refugia managed by other means, such as mowing, might be a prudent alternative to burning (Swengel and Swengel 2007).
- Maintain secure fire lines (fuel breaks) around the perimeter of a preserve but minimize the use of plowed or disked fire lines within preserve interiors. Besides favoring invasive species, plowed fire lines disrupt hydrology and serve as movement barriers or

death traps to some small animals, among other problems (Noss 2018). Alternatives to aggressively plowed or disked fire lines include use of natural fire-proof features such as streams or lakes, use of roads, and mowing (ideally followed by burning to provide fire-proof “black lines”).

- Develop a detailed fire history map for each preserve to guide planning of prescribed burns.
- Update burn unit boundaries and plans regularly based on data from previous burns (i.e., apply adaptive management).
- Suppress wildfires only when they pose a risk to adjacent or nearby properties. When possible, allow managed wildfire, which has shown many ecological benefits (Noss et al. 2006; Ryan et al. 2013; North et al. 2015). Ecologists and land managers increasingly view managed wildfire as beneficial in reducing the fire deficit in the United States, albeit this view is more prevalent in the West and far North and is only beginning to develop elsewhere in North America.
- An example of guidelines for fire management and immediate post-fire rehabilitation for natural areas is provided by Safford and Wright (2015).

Visitor Management

Many protected natural areas under various jurisdictions are open to the public for a variety of recreational uses, whereas some are restricted to passive or nonconsumptive uses, and others are open only to permitted scientific research. For natural areas open to compatible recreation, the following **best management practices** are appropriate:

- Perhaps the best direct human use of natural areas is increasing the level of public appreciation of nature. Therefore, education about the natural environment is the highest priority for visitor management in a preserve.
- Trails, kiosks, and interpretive signage will introduce visitors to the natural communities of the preserves, their characteristic disturbance regimes (e.g., fire), flora, fauna, and other aspects of natural history.
- Avoid placing trails in close proximity to highly sensitive natural features such as very rare plants or wading bird rookeries.
- A growing body of literature is showing that even nonmotorized recreation (hiking, biking, dog-walking) can have measurable impacts on wildlife use (Reed and Merenlender 2008; Larson et al. 2019; Dertien et al. 2021). Where possible, natural areas managers should regulate visitor use to mitigate undesirable impacts to natural area objectives.

Human Diversity, Inclusivity, and Equity

People of European origin have dominated the natural areas movement in North America throughout its history (Nijhuis 2021). Leaders have been mostly white, male, and financially secure, although there have been exceptions; important historical gender exceptions were E. Lucy Braun and Helen T. Gaige, who participated in Shelford's ESA Committee on the Preservation of Natural Conditions. If the public is going to continue to support natural areas, for example by voting for tax increases that supply funding for natural areas acquisition and management, then natural areas must be accessible and appealing to all classes of people, including inner-city dwellers. What kinds of actions are needed to increase involvement of minorities and underrepresented groups in the natural areas movement?

- Wherever possible, engage deeply with Indigenous people in the co-management of natural areas, as their long and deep knowledge of ecosystems and species within and across their ancestral homelands will often prove invaluable to their conservation and potential restoration for both ecological and cultural benefits; this also may improve relationships between Indigenous and non-indigenous peoples.
- Locate protected natural areas (or seminatural areas, if this is the only option) near neighborhoods dominated by underrepresented groups. There is no substitute for direct contact with nature, ideally starting early in life, for encouraging positive attitudes toward natural areas.
- Provide tangible incentives, such as scholarships, for young people from underrepresented groups to receive education and training in ecology, conservation biology, natural areas management and restoration, and related fields.
- Significantly increase salaries for various positions within the natural areas profession to ensure that the profession of natural areas management appeals to people from all socio-economic backgrounds.

Conclusions: Lessons for Success in the 21st Century

> "History ends in ecology, or nothing" (Rowe 1990)

Our collective survival depends upon nature and its cornerstone natural areas, writ large. We need to remember this as we continue to recognize and steward our natural areas. Canadian ecologist Stan Rowe considered the significance of natural areas as a "needed rallying point for earth care" (Rowe 1976). Resilient landscapes at scale, whether at home or globally, have natural areas at their core, delivering the habitats, the fellow species, and the water, energy, and carbon cycles, which alone can keep us whole. Indigenous peoples have long recognized the strong connection between humans and the natural world, particularly for the overall well-being of their communities. Non-indigenous peoples could stand to learn from Indigenous partners, particularly through co-management of the natural areas within their ancestral homelands.

As we acknowledged earlier, none of the current or foreseeable future challenges to natural areas addressed in this paper are completely new. The magnitude of these challenges is, however, becoming unprecedented and increasingly urgent. Most profoundly, we now recognize that anthropogenic climate change is an existential threat to both human civilization and nature. Given these major threats, important lessons emerge from our research and experiences and our understanding of the values of natural areas. We summarize these lessons below.

First, we should not rush to discard the values and norms that mobilized the natural areas movement through the 20th century and remain prominent today. Essentially these values all are still relevant and true. Many recent criticisms of natural areas preservation are caricatures of the movement. Few, if any, natural areas professionals ever truly believed that you could simply put a fence around an area, walk away, and it would remain in that condition in perpetuity. Virtually all ecologists, throughout the history of the field, have been keenly aware of the dynamic nature of ecosystems (McIntosh 1985). The suggestion that "generally, conservation aims to reduce or prevent both abiotic and biotic change" (Hobbs et al. 2009) is a gross oversimplification. Few, if any, conservationists seek to prevent ecological change. Indeed, the field of restoration ecology actively seeks to change degraded ecosystems. And most conservationists probably would agree that evolutionary change, such as improved adaptation to changing climate, is highly desirable.

Awareness of the dynamism of nature has grown, however, in concert with improvements in our understanding of disturbance ecology and observations of the impacts of climate change. Ecosystems are changing faster today than over most of the previous century, and often in seemingly novel directions. This new level of awareness of environmental change and the dynamic nature of ecosystems should stimulate questions about some long-cherished assumptions about natural areas conservation, restoration, and management. Questioning assumptions does not, however, mean abandoning fundamental values and goals. It might instead mean that we need to develop better forecasting skills and more rigorously factor timing into our planning.

Second, as environmental change accelerates, the value of natural areas as benchmarks (assuming they remain in less degraded condition than the surrounding landscape) increases, as does their role in safeguarding biodiversity and ecological integrity. Novel ecosystems are already emerging inside and outside of natural areas, and they are not devoid of conservation value (Hobbs et al. 2009). Recognizing the conservation value of "historic, hybrid, and novel ecosystems" (Hobbs et al. 2014) is consistent with the resist change, accept change, or guide change options for addressing climate change (Aplet and McKinley 2017; Jackson 2021; Lynch et al. 2021), as well as with the increased recognition of ecosystem transformation as an adaptive framework, as discussed earlier.

We always need to remember that “natural” is a relative concept and that a spectrum of naturalness and wildness exists in virtually all landscapes (Aplet 1999). Arbitrarily focusing on just the most pristine portion of the naturalness gradient and ignoring the rest would be a mistake. Semi-natural landscapes such as mixed or “semi-improved” pastures in Florida, for example, could be considered either hybrid or novel ecosystems. Yet they have significant conservation values, including serving as preferred habitat for some of Florida’s birds of greatest conservation concern (Morrison and Humphrey 2001). These include the state-Threatened Florida sandhill crane (*Antigone canadensis pratensis*), the federally Threatened crested caracara (*Caracara cheriway*), and the state-Threatened Florida burrowing owl (*Athene cunicularia floridana*). The latter two species presumably migrated to Florida from western North America during the Pliocene or Pleistocene along the Gulf Coastal Corridor and probably once depended on grasslands grazed by now extinct megaherbivores (Noss 2013). Ironically, these novel or hybrid ranchland ecosystems may mimic deep historical ecosystems! Natural areas that retain examples of more recent pre-settlement-type ecosystems provide a scientifically valuable comparison to these ranchlands. Other examples of novel or anthropogenic ecosystems playing valuable conservation roles include the *Eucalyptus* groves of California (but see Griffiths and Villablanca 2015) and the species-rich chalk grasslands grazed by sheep in southern England and continental Europe (Duffey et al. 1974).

Third, one major development in ecology and conservation biology in the late 20th and early 21st centuries is increased recognition of landscape ecology. Large natural areas are landscapes in themselves, but they are still influenced by activities and processes in the larger landscape that surrounds them. In many regions, such as southeastern Canada and the U.S. Midwest, most natural areas are small sites embedded in human-dominated landscapes; some of these natural areas comprise single natural communities. The effects of the surrounding landscape are more profound for these small natural areas, due to edge effects and other processes (Laurance and Yensen 1991; Murcia 1995). Edge effects vary in intensity and impacts according to matrix characteristics and many other factors, which are still inadequately studied across a range of North American landscapes. Natural areas managers, where possible, should work with land-use planners to improve the landscape context surrounding natural areas. Expanding the size of reserves to mitigate deleterious edge effects may be possible in some cases.

Fourth, conflicts between species-level and ecosystem-level management remain problematic today. Most natural areas managers are aware that both species and ecosystems deserve conservation attention. Because the needs of individual species sometimes conflict, managing for ecosystems seems a sensible way to reduce disputes (Noss 1996). Especially in regions with many conservation-reliant species, there are only so many species that we can conserve or manage individually without being overwhelmed. The biological status of species is usually linked directly to the condition of the ecosystems with which they are associated. Protecting and managing ecosystems is therefore a cost-efficient way to protect multiple species with shared biological needs and shared threats (Noss et al. 2021).

On the other hand, among the best indicators of the quality or integrity of ecosystems is the presence and viability of species that are characteristic of that ecosystem. Hence, species-based indices such as the Floristic Quality Index (FQI) are used to assess the quality and conservation importance of natural areas (Wilhelm 1977). Moreover, foundation species, apex predators, ecological engineers, and other strongly interacting species commonly control the structure and diversity of the ecosystem (Soulé et al. 2003, 2005); these species must be maintained in ecologically functional, not just minimally viable, populations. Some species demand individual attention because they are so highly imperiled that they would perish without it. It is inescapable that natural areas managers must attend to at least some individual species as well as to the ecosystems in which they occur. This is entirely consistent with the stated goal of the U.S. Endangered Species Act "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved" (P.L. 94-325, as amended).

The following performance measures, along with an ecological monitoring program, will measure the ecological success of natural areas conservation, restoration, and management:

- Increases in cover of native plants and the floristic quality of natural communities and declines in cover of nonnative and invasive plants and communities;
- Proportions of natural community types across the landscape moving toward estimated desired range of conditions in appropriate terrain/habitat locations;
- Long-term persistence of at-risk species and as many other native species as possible;
- Populations of at-risk species recovering or fluctuating over time within an acceptable range of variation;
- Fire regimes moving toward ranges of variation (in fire-return interval, severity, seasonality, and other components of the fire regime) that are compatible with the ecological communities and ecosystem processes that are desired on the target landscape; and
- Human uses that are compatible with biological conservation to the greatest degree possible.

In closing, we offer provisional answers to the questions posed at the beginning of this report.

1. *Are natural areas still relevant to the public in the 21st century?* Yes, given the increasing human population and the well-documented benefits of exposure to nature for

human physical, intellectual, and emotional well-being, natural areas are more relevant than ever. That said, many people are not aware of these benefits and therefore do not perceive the relevance of natural areas to their personal lives. More education and more direct experience in natural areas are needed across the geographic, economic, ethnic, and sociocultural spectrum. Natural areas professionals need to actively engage with all kinds of people.

2. *Do natural areas still serve the purposes for which they were established?* The answer is context specific. Many natural areas do still serve these purposes, but others have been degraded to the point that their contributions to conservation purposes have been diminished. Restoration is needed where it is possible, but we must recognize that some small natural areas embedded in intensively used (e.g., urban) landscapes may be sacrifice zones from a conventional natural areas perspective, yet may still be of recreational or other value to people.
3. *Have the values (real and perceived) of natural areas changed over time?* We believe that the fundamental values of natural areas, as reviewed in this report—as benchmarks, as protectors of biodiversity, as habitat for rare species, etc.—have not changed. What has changed is that these values are more challenging to maintain in a more crowded and rapidly changing world.
4. *How might natural areas be better designed, managed, and marketed to meet changing environmental and social conditions over the remainder of this century?* The design and management needs and challenges of natural areas are reviewed extensively in this report. We strongly recommend further research and synthesis to understand how natural areas can be more effectively marketed to a broad public and managed to meet diverse social needs. To repeat a statement from the introduction to this report: *Making the natural areas profession—and the appreciation of natural areas—more diverse and inclusive is not only ethically correct but may be essential to the survival of natural areas as a public good through this century and beyond.*

Literature Cited

Abrahms, B., D. DiPietro, A. Graffis, and A. Hollander. 2017. Managing biodiversity under climate change: Challenges, frameworks, and tools for adaptation. Biodiversity and Conservation 26:2277–2293.

Ackerly, D.D., S.R. Loarie, W.K. Cornwell, S.B. Weiss, H. Hamilton, R. Branciforte, and N.J.B. Kraft. 2010. The geography of climate change: Implications for conservation biogeography. Diversity and Distributions 16:476–487.

Adger, W.N., J. Barnett, K. Brown, N. Marshall, and K. O'Brien. 2013. Cultural dimensions of climate change impacts and adaptation. Nature Climate Change 3:112–117.

Agee, J.K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC.

Albano, C.M. 2015. Identification of geophysically diverse locations that may facilitate species' persistence and adaptation to climate change in the southwestern United States. Landscape Ecology 30:1023–1037.

Allen, C.R., J.J. Fontaine, K.L. Pope, and A.S. Garmestani. 2011. Adaptive management for a turbulent future. Journal of Environmental Management 92:1339–1345.

[AAAS] American Association for the Advancement of Science. 1963. Natural Areas as Research Facilities. AAAS, Washington, DC.

Anderson M.G., and C.E. Ferree. 2010. Conserving the stage: Climate change and the geophysical underpinnings of species diversity. PLOS One 5(7):e11554.

Anderson, M., P.J. Comer, P. Beier, J.J. Lawler, C.A. Schloss, S. Buttrick, C.M. Albano, and D.P. Faith. 2015. Case studies of conservation plans that incorporate geodiversity. Conservation Biology 29:680–691.

Andreasen, J.K., R.V. O'Neill, R. Noss, and N.C. Slosser. 2001. Considerations for the development of a terrestrial index of ecological integrity. Ecological Indicators 1:21–35.

Aplet, G. 1999. On the nature of wildness: Exploring what wilderness really protects. University of Denver Law Review 76:347–367.

Aplet, G.H., and D.N. Cole. 2010. The trouble with naturalness: Rethinking park and wilderness goals. Pp. 12–29 in D.N. Cole and L. Yung, eds., Beyond Naturalness. Island Press, Washington, DC.

Aplet, G.H., and W.K. Keeton. 1999. Application of historical range of variability concepts to biodiversity conservation. Pp. 71–86 in R.K. Baydack, H. Campa III, and J.B. Haufler, eds., Practical Approaches to the Conservation of Biological Diversity. Island Press, Covelo, CA.

Aplet, G.H., and P.S. McKinley. 2017. A portfolio approach to managing ecological risks of global change. Ecosystem Health and Sustainability 3(2):e01261.

Baldwin, R.F., S.C. Trombulak, P.B. Leonard, R.F. Noss, J.A. Hilty, H.P. Possingham, L. Scarlett, and M.G. Anderson. 2018. The future of landscape conservation. BioScience 68:60–63.

Barrett, S.W., T. DeMeo, J.L. Jones, J.D. Zeiler, and L.C. Hutter. 2006. Assessing ecological departure from reference conditions with the Fire Regime Condition Class (FRCC) mapping tool. Pp. 575–585 in P.L. Andrews and B.W. Butler, compilers. Fuels Management: How to Measure Success. 28–30 March 2006; Portland, OR. Proceedings RMRS-P-41, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Batllori, E., C. Miller, M.A. Parisien, S.A. Parks, and M.A. Moritz. 2014. Is U.S. climatic diversity well represented within the existing federal protection network? Ecological Applications 24:1898–1907.

Bátori, Z., A. Vojtkó, T. Farkas, A. Szabó, K. Havadtoi, A.E. Vojtkó, C. Tölgyesi, V. Cseh, L. Erdos, I.E. Maák, and G. Keppel. 2017. Large- and small-scale environmental factors drive distributions of cool-adapted plants in karstic microrefugia. Annals of Botany 119:301–309.

Beckage, B., W.J. Platt, and L.J. Gross. 2009. Vegetation, fire, and feedbacks: A disturbance-mediated model of savannas. American Naturalist 174:805–818.

Beier, P., and B. Brost. 2010. Use of land facets to plan for climate change: Conserving the arenas, not the actors. Conservation Biology 24:701–710.

Belote, R.T., M.S. Dietz, C.N. Jenkins, P.S. McKinley, G.H. Irwin, T.J. Fullman, J.C. Leppi, and G.H. Aplet. 2017a. Wild, connected, and diverse: Building a more resilient system of protected areas. Ecological Applications 27:1050–1056.

Belote, R.T., M.S. Dietz, P.S. McKinley, A.A. Carson, C. Carroll, C.N. Jenkins, D.L. Urban, T.J. Fullman, J.C. Leppi, and G.H. Aplet. 2017b. Mapping conservation strategies under a changing climate. BioScience 67:494–497.

Bradley, B.A., C.A. Curtis, E.J. Fusco, J.T. Abatzoglou, J.K. Balch, S. Dadashi, and M. Tuanmu. 2018. Cheatgrass (Bromus tectorum) distribution in the intermountain Western United States and its relationship to fire frequency, seasonality, and ignitions. Biological Invasions 20:1493–1506.

Brockway, D.G., and K.W. Outcalt. 2000. Restoring longleaf pine wiregrass ecosystems: Hexazinone application enhances effects of prescribed fire. Forest Ecology and Management 137:121–138.

Brodie, J.F., S. Lieberman, A. Moehrenschlager, K.H. Redford, J.P. Rodriguez, M. Schwartz, P.J. Seddon, and J.E.M. Watson. 2021. Global policy for assisted colonization of species. Science 372:456–458.

Buckley, Y.M., and Y. Han. 2014. Managing the side effects of invasion control. Science 344:975–976.

Butt, N., H.P. Possingham, C. De Los Rios, R. Maggini, R.A. Fuller, S.L. Maxwell, and J.E.M. Watson. 2016. Challenges in assessing the vulnerability of species to climate change to inform conservation actions. Biological Conservation 199:10–15.

Cain, S.A. 1947. Characteristics of natural areas and factors in their development. Ecological Monographs 17:185–200.

Canadell, J.G., D. Pataki, and L. Pitelka, eds. 2007. Terrestrial Ecosystems in a Changing World. The IGBP Series, Springer-Verlag, Berlin and Heidelberg, Germany.

Casazza, M.L., C.T. Overton, T.V.D. Bui, J.M. Hull, J.D. Albertson, V.K. Bloom, S. Bobzien, J. McBroom, M. Latta, P. Olofson, et al. 2016. Endangered species management and ecosystem restoration: Finding the common ground. Ecology and Society 21(1):19.

Ceballos, G., P.R. Ehrlich, A.D. Barnosky, A. Garcia, R.M. Pringle, and T.M. Palmer. 2015. Accelerated modern human-induced species losses: Entering the sixth mass extinction. Science Advances e1400253.

Cheng, S. 2004. Forest Service Research Natural Areas in California. PSW-GTR-188, USDA Forest Service, Pacific Southwest Research Station, Berkeley, CA.

Clark, J.A., and E. Harvey. 2002. Assessing multi-species recovery plans under the Endangered Species Act. Ecological Applications 12:655–662.

Cole, D.N., and L. Yung, eds. 2010. Beyond Naturalness: Rethinking Park and Wilderness Stewardship in an Era of Rapid Change. Island Press, Washington, DC.

Colloff, M.J., S. Lavorel, L.E. van Kerkhoff, C.A. Wyborn, I. Fazey, R. Gorddard, G.M. Mace, W.B. Foden, M. Dunlop, I.C. Prentice, et al. 2017. Transforming conservation science and practice for a postnormal world. Conservation Biology 31:1008–1017.

Comer, P.J., J.C. Hak, C. Josse, and R. Smyth. 2020. Long-term loss in extent and current protection of terrestrial ecosystem diversity in the temperate and tropical Americas. PLOS One 15(6):e0234960.

Comer, P.J., J.C. Hak, and E. Seddon. 2022. Documenting at-risk status of terrestrial ecosystems in temperate and tropical North America. Conservation Science and Practice 4:e603.

[CBD] Convention on Biological Diversity. 2018. Protected Areas and Other Effective Area-Based Conservation Measures (Decision 14/8). Convention on Biological Diversity, Montreal, QC.

Coppoletta, M., H.D. Safford, B.L. Estes, M.D. Meyer, S.E. Gross, K.E. Merriam, R.J. Butz, and N.A. Molinari. 2019. Fire regime alteration in natural areas underscores the need to restore a key ecological process. Natural Areas Journal 39:250–263.

Covington, W.W., and M.M. Moore. 1994. Southwestern ponderosa pine forest structure and resource conditions: Changes since Euro-American settlement. Journal of Forestry 92:39–47.

Croker, R.A. 1991. Pioneer Ecologist: The Life and Work of Victor Ernest Shelford 1877–1968. Smithsonian Institution Press, Washington, DC.

Crooks, K.R., and M.E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563–566.

Cross, M.S., E.S. Zavaleta, D. Bachelet, M.L. Brooks, C.A.F. Enquist, E. Fleishman, L.J. Graumlich, C.R. Groves, L. Hannah, L. Hansen, et al. 2012. The Adaptation for Conservation Targets (ACT) framework: A tool for incorporating climate change into natural resource management. Environmental Management 50:341–351.

D'Antonio, C., and S.L. Flory. 2017. Long-term dynamics and impacts of plant invasions. Journal of Ecology 105:1459–1461.

Davies, K.W., E.A. Leger, C.S. Boyd, and L.M. Hallett. 2021. Living with exotic annual grasses in the sagebrush ecosystem. Journal of Environmental Management 288:112417.

Davies, K.W., and A.M. Nafus. 2013. Exotic annual grass invasion alters fuel amounts, continuity, and moisture content. International Journal of Wildland Fire 22:353–358.

Davis, M.A., M.K. Chew, R.J. Hobbs, A.E. Lugo, J.J. Ewel, G.J. Vermeij, J.H. Brown, M.L. Rosenzweig, M.R. Gardener, S.P. Carroll, et al. 2011. Don't judge species on their origins. Nature 474:153–154.

Dertien, J.S., C. Larson, and S.E. Reed. 2021. Recreation effects on wildlife: A review of potential quantitative thresholds. Nature Conservation 44:51–68.

Dobrowski, S.Z. 2011. A climatic basis for microrefugia: The influence of terrain on climate. Global Change Biology 17:1022–1035.

Duffey, E., M.G. Morris, J. Sheail, L.K. Ward, D.A. Wells, and T.C.E. Wells. 1974. Grassland Ecology and Wildlife Management. Chapman and Hall, London, UK.

Dukes, J.S., and H.A. Mooney. 1999. Does global change increase the success of biological invaders? Trends in Ecology and Evolution 14:135–139.

Eagles, P.F.J. 1984. The Planning and Management of Environmentally Sensitive Areas. Longman, London and New York.

Enquist, C.A.F., and S.T. Jackson. 2016. Ensuring Coordination Among Regional Climate Science Programs, National Adaptation Forum, St. Louis, MO, 14 May 2015. Eos, Meeting

Reports, 25 January 2016.

Federation of Ontario Naturalists. 1934. Sanctuaries and the Preservation of Wild Life in Ontario. Pub. 2, Federation of Ontario Naturalists.

Fell, G.B. 1983. The natural area movement in the United States, its past and its future. Natural Areas Journal 3(4):47–55.

Fill, J.M., W.J. Platt, S.M. Welch, J.L. Waldron, and T.A. Mousseau. 2015. Updating models for restoration and management of fiery ecosystems. Forest Ecology and Management 356:54–63.

Fletcher, R.J., R.K. Didham, C. Banks-Leite, J. Barlow, R.M. Ewers, J. Rosindell, R.D. Holt, A. Gonzalez, R. Pardini, E.I. Damschen, et al. 2018. Is habitat fragmentation good for biodiversity? Biological Conservation 226:9–15.

Flies, E.J., C. Skelly, S. Singh Negi, P. Proabhakaran, Q. Liu, K. Liu, F.C. Goldizen, C. Lease, and P. Weinstein. 2017. Biodiverse green spaces: A prescription for global urban health. Frontiers in Ecology and the Environment 15:510–516.

Flory, S.L., J. Bauer, R.P. Phillips, and K. Clay. 2017. Effects of a non-native grass invasion decline over time. Journal of Ecology 105:1475–1484.

Forman, R.T.T., and M. Godron. 1981. Patches and structural components for a landscape ecology. BioScience 31:733–740.

Fougères D., M. Jones, P.D. McElwee, A. Andrade, and S.R. Edwards. 2022. Transformative conservation of ecosystems. Global Sustainability 5:e5.

Franklin, J.F. 1993. Preserving biodiversity: Species, ecosystems, or landscapes. Ecological Applications 3:202–205.

Franklin, J.F., R.E. Jenkins, and R.M. Romancier. 1972. Research Natural Areas: Contributors to environmental quality programs. Journal of Environmental Quality 1:133–139.

Franklin, J.F., K.N. Johnson, and D.L. Johnson. 2018. Ecological Forest Management. Waveland Press, Long Grove, IL.

Franklin, J.F., and J.M. Trappe. 1968. Natural areas: Needs, concepts, and criteria. Journal of Forestry 66:456–461.

Freedman, B. 2013. A History of The Nature Conservancy of Canada. Oxford University Press, Don Mills, ON.

Friesen, L.E., P.F.J. Eagles, and R.J. MacKay. 1995. Effects of residential development on forest-dwelling Neotropical migrant songbirds. Conservation Biology 9:1408–1414.

Gentry, A.W. 1986. Endemism in tropical versus temperate plant communities. Pp. 153–181 in M.E. Soulé, ed., Conservation Biology: The Science of Scarcity and Diversity. Sinauer, Sunderland, MA.

Glick, P., B.A. Stein, and N.A. Edelson, eds. 2011. Scanning the Conservation Horizon: A Guide to Climate Change Vulnerability Assessment. National Wildlife Federation, Washington, DC.

Greenberg, C.H., and Collins, B. eds., 2021. Fire Ecology and Management: Past, Present, and Future of US Forested Ecosystems. Springer, Cham, Switzerland.

Griffiths, J., and F. Villablanca. 2015. Managing monarch butterfly overwintering groves: Making room among the eucalyptus. California Fish and Game 101:40–50.

Groom, M.J., and C.H. Vynne. 2006. Habitat degradation and loss. Pp. 173–212 in M.J. Groom, G.K. Meffe, and R.C. Carroll, eds., Principles of Conservation Biology. 3rd ed. Sinauer Associates, Sunderland, MA.

Groves, C. 2003. Drafting a Conservation Blueprint: A Practitioner's Guide to Planning for Biodiversity. Island Press, Washington, DC.

Groves, C.R., and E.T. Game. 2015. Conservation Planning: Informed Decisions for a Healthier Planet. W.H. Freeman, New York, NY.

Groves, C.R., E.T. Game, M.G. Anderson, M. Cross, C. Enquist, Z. Ferdaña, E. Girvetz, A. Gondor, K.R. Hall, J. Higgins, et al. 2012. Incorporating climate change into systematic conservation planning. Biodiversity and Conservation 21:1651–1671.

Groves, C.R., D.B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, and M.G. Anderson. 2002. Planning for biodiversity conservation: Putting conservation science into practice. BioScience 52:499–512.

Groves, C., L. Valutis, D. Vosick, B. Neely, K. Wheaton, J. Touval, and B. Runnels. 2000. Designing a Geography of Hope: A Practitioner's Handbook for Ecoregional Conservation Planning. The Nature Conservancy, Arlington, VA.

Guelph Conference. 1942. Conservation and post-war rehabilitation. Toronto, ON.

Gunderson, L.H., and C.S. Holling, eds. 2002. Panarchy: Understanding Transformations in Human and Natural Systems. Island Press, Washington, DC.

Haddad, N.M., L.A. Brudvig, J. Clobert, K.F. Davies, A. Gonzalez, R.D. Holt, T.E. Lovejoy, J.O. Sexton, M.P. Austin, C.D. Collins, et al. 2015. Habitat fragmentation and its lasting impact on Earth. Science Advances 1:e1500052.

Hagmann, R.K., P.F. Hessburg, S.J. Prichard, N.A. Povak, P.M. Brown, P.Z. Fulé, R.E. Keane, E.E. Knapp, J.M. Lydersen, K.L. Metlen, et al. 2021. Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests. Ecological Applications 31:e02431.

Hanski, I. 1998. Metapopulation dynamics. Nature 396:41–49.

Harper, R.M. 1911. The relation of climax vegetation to islands and peninsulas. Bulletin of the Torrey Botanical Club 38:515–525.

Harris, L.D. 1984. The Fragmented Forest: Island Biogeography Theory and the Preservation of Biotic Diversity. University of Chicago Press, Chicago, IL.

Harrison, S., and R. Noss. 2017. Endemism hotspots are linked to stable climatic refugia. Annals of Botany 119:207–214.

Heller, N.E., and E.S. Zavaleta. 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation 142:14–32.

Hellmann, J.J., J.E. Byers, B.G. Bierwagen, and J.S. Dukes. 2008. Five potential consequences of climate change for invasive species. Conservation Biology 22:534–543.

Hilty, J.A., W.Z. Lidicker Jr., and A.M. Merenlender. 2006. Corridor Ecology: The Science and

Practice of Linking Landscapes for Biodiversity Conservation. Island Press, Washington, DC.

Hobbs, R.J., E. Higgs, and J.A. Harris. 2009. Novel ecosystems: Implications for conservation and restoration. Trends in Ecology and Evolution 24:599–605.

Hobbs, R.J., E. Higgs, C.M. Hall, P. Bridgewater, F.S. Chapin III, E.C. Ellis, J.J. Ewel, L.M. Hallett, J. Harris, K.B. Hulvey, et al. 2014. Managing the whole landscape: Historical, hybrid, and novel ecosystem. Frontiers in Ecology and the Environment 12:557–564.

Howes, A.L., M. Maron, and C.A. McAlpine. 2010. Bayesian networks and adaptive management of wildlife habitat. Conservation Biology 24:974–983.

Hulme, P.E., P. Pyšek, V. Jarošik, U. Schaffner, and M. Vilà. 2014. Greater focus needed on alien plant impacts in protected areas. Conservation Letters 7:459–466.

Humphrey, L.D., and E.W. Schupp. 2004. Competition as a barrier to establishment of a native perennial grass (Elmus elymoides) in alien annual grass (Bromus tectorum) communities. Journal of Arid Environments 58:405–422.

Hylander, K., C. Greiser, D.M. Christiansen, and I.A. Koelemeijer. 2022. Climate adaptation of biodiversity conservation in managed forest landscapes. Conservation Biology 36:e13847.

Iffrig, G.F. 1981. The Natural Areas Association – A brief history of formation. Journal of the Natural Areas Association 1(1):1–2.

[IPCC] Intergovernmental Panel on Climate Change. 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, et al., eds. Cambridge University Press, Cambridge, UK, and New York, NY, USA. doi:10.1017/9781009157896.

Jackson, S.T. 2021. Transformational ecology and climate change. Science 373:1085–1087.

Jansson R. 2003. Global patterns in endemism explained by past climatic change. Proceedings of the Royal Society B: Biological Sciences 270:583–590.

Jenkins, R.E. 1978. Heritage classification: The elements of ecological diversity. Nature Conservancy News 38(1):24–25, 30.

Jenkins, R.E. 1985. Information methods: Why the heritage programs work. Nature Conservancy News 35(6):21–23.

Johnston, A., M. Ausden, A.M. Dodd, R.B. Bradbury, D.E. Chamberlain, F. Jiguet, C.D. Thomas, A.S.C.P. Cook, S.E. Newson, N. Ockendon, et al. 2013. Observed and predicted effects of climate change on species abundance in protected areas. Nature Climate Change 3:1055–1061.

Karr, J.R., and E.W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington DC.

Karr, J.R., E.R. Larson, and E.W. Chu. 2022. Ecological integrity is both real and valuable. Conservation Science and Practice 4(2):e583.

Keeley, J.E., J.G. Pausas, P.W. Rundel, W.J. Bond, and R.A. Bradstock. 2011. Fire as an evolutionary pressure shaping plant traits. Trends in Plant Science 16:406–411.

Kellert, S.R. 2002. Experiencing nature: Affective, cognitive, and evaluative development in children. Pp. 117–151 in P.H. Kahn and S.R. Kellert, eds., Children and Nature: Psychological, Sociocultural, and Evolutionary Investigations. MIT Press, Cambridge, MA.

Kendeigh, S.C., H.I. Baldwin, V.H. Cahalane, C.H.D. Clarke, C. Cottam, W.P. Cottam, I. McT. Cowan, P. Dansereau, J.H. Davis, F.W. Emerson, et al. 1950–1951. Nature sanctuaries in the United States and Canada: A preliminary inventory. The Living Wilderness 15(35):1–45.

Keppel, G., K.P. Van Niel, G.W. Wardell-Johnson, C.J. Yates, M. Byrne, L. Mucina, A.G.T. Schut, S.D. Hopper, and S.E. Franklin. 2012. Refugia: Identifying and understanding safe havens for biodiversity under climate change. Global Ecology and Biogeography 21:393–404.

Keystone Center. 1991. Final Consensus Report of the Keystone Policy Dialogue on Biological Diversity on Federal Lands. The Keystone Center, Keystone, CO.

Killan, G. 1993. Protected Places: A History of Ontario's Provincial Parks System. Dundurn Press, Toronto, ON.

Knapp, P.A. 1995. Intermountain West lightning-caused fires: Climatic predictors of area burned. Journal of Range Management 48:85–91.

Knapp, W.M., A. Frances, R. Noss, R.F.C. Naczi, A. Weakley, G.D. Gann, B.G. Baldwin, J. Miller, P. McIntyre, B.D. Mishler, et al. 2021. Analysis of the extinct plants of North America north of Mexico provides a baseline for the Anthropocene. Conservation Biology 35:360–368.

Kooyman, R., and M. Rossetto. 2008. Definition of plant functional groups for informing implementation scenarios in resource-limited multi-species recovery planning. Biodiversity and Conservation 17:2917–2937.

Kukkala, A.S., and A. Moilanen. 2013. Core concepts of spatial prioritization in systematic conservation planning. Biological Reviews 88:443–464.

Kunin, W.E., and K.J. Gaston. 1993. The biology of rarity: Patterns, causes, and consequences. Trends in Ecology and Evolution 8:298–301.

Lam, D., E. Hinz, D. Lang, M. Tengö, H. von Wehrden, and B. Martín-López. 2020. Indigenous and local knowledge in sustainability transformations research: A literature review. Ecology and Society 25(1):3.

Larson, C., S.E. Reed, A. Merenlender, and K.R. Crooks. 2019. A meta-analysis of recreation effects on vertebrate species richness and abundance. Conservation Science and Practice 1(10):e93.

Laurance, W.E., and E. Yensen. 1991. Predicting the impacts of edge effects in fragmented habitats. Biological Conservation 55:77–92.

Leger, E.A., and E.M. Goergen. 2017. Invasive Bromus tectorum alters natural selection in arid systems. Journal of Ecology 105:1509–1520.

Lehmann, P., T. Ammunét, M. Barton, A. Battisti, S.D. Eigenbrode, J.U. Jepsen, G. Kalinkat, S. Neuvonen, P. Niemelä, J.S. Terblanche, et al. 2020. Complex responses of global insect pests to climate warming. Frontiers in Ecology and the Environment 18:141–150.

Leopold, A. 1949. A Sand County Almanac. Oxford University Press, New York, NY.

Lesica, P., and F.W. Allendorf. 1995. When are peripheral populations valuable for conservation? Conservation Biology 9:753–760.

Lindenmayer, D.B., and J.F. Franklin. 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, DC.

Long, J.W., F.K. Lake, R.W. Goode, and B.M. Burnette. 2020. How traditional tribal perspectives influence ecosystem restoration. Ecopsychology 12:1–12.

Louv, R. 2011. The Nature Principle: Human Restoration and the End of Nature-Deficit Disorder. Algonquin Books, Chapel Hill, NC.

Lovejoy, T.E. 1980. Foreword. Pp. ix–x in M.E. Soulé and B.A. Wilcox, eds., Conservation Biology: An Evolutionary-Ecological Perspective. Sinauer, Sunderland, MA.

Lynch, A.J., L.M. Thompson, E.A. Beever, D.N. Cole, A.C. Engman, C. Hawkins Hoffman, S.T. Jackson, T.J. Krabbenhoft, D.J. Lawrence, D. Limpinsel, et al. 2021. Managing for RADical ecosystem change: Applying the Resist-Adapt-Direct (RAD) framework. Frontiers in Ecology and the Environment 19:461–469.

Main, D. 2021. Florida enacts sweeping law to protect its wildlife corridors. National Geographic, 30 June 2021. https://www.nationalgeographic.com/animals/article/florida-wildlife-corridor-legislation-unanimous-environmental-law

Margules, C.R., and R.L. Pressey. 2000. Systematic conservation planning. Nature 405:243–253.

Martin, R.E., and D.B. Sapsis. 1992. Fires as agents of biodiversity: Pyrodiversity promotes biodiversity. Pp. 150–157 in H.M. Kerner, ed., Proceedings of the Symposium on Biodiversity in Northwestern California, 1991. Wildland Resources Center, University of California, Berkeley, CA.

Massie, M.H., T.M. Wilson, A.T. Morzillo, and E. B. Henderson. 2016. Natural areas as a basis for assessing ecosystem vulnerability to climate change. Ecosphere 7(11):e01563.

McCallum, M.L. 2015. Vertebrate biodiversity losses point to a sixth mass extinction. Biodiversity and Conservation 24:2497–2519.

McIntosh, R.P. 1985. The Background of Ecology: Concept and Theory. Cambridge University Press, Cambridge, UK.

Menges, E.S. 2007. Integrating demography and fire management: An example from Florida scrub. Australian Journal of Botany 55:261–272.

Menges, E.S., and D.R. Gordon. 2010. Should mechanical treatments and herbicides be used as fire surrogates to manage Florida's uplands? A review. Florida Scientist 73:147–174.

Millar, C.I., N.L. Stephenson, and S.L. Stephens. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications 17:2145–2151.

Millar, C.I., and N.L. Stephenson. 2015. Temperate forest health in an era of emerging megadisturbance. Science 349:823–826

Miller, P., and W.E. Rees. 2000. Introduction. Pp. 3–18 in D. Pimentel, L. Westra, and R. Noss, eds., Ecological Integrity: Integrating Environment, Conservation, and Health. Island Press, Washington, DC.

Mitchell, R.J., Y. Liu, J.J. O'Brien, K.J. Elliott, G. Starr, C.F. Miniat, and J.K. Hiers. 2014. Future climate and fire interactions in the southeastern region of the United States. Forest Ecology and Management 327:316–326.

Mittermeier, R.A., W.R. Turner, F.W. Larsen, T.W. Brooks, and C. Gascon. 2011. Global biodiversity conservation: The critical role of hotspots. Pp. 3–22 in F.E. Zachos and J.C. Habel, eds., Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas. Springer Verlag, Heidelberg, Germany.

Moir, W.H. 1972. Natural areas. Science 177:396–400.

Moore, J.W., and D.E. Schindler. 2022. Getting ahead of climate change for ecological adaptation and resilience. Science 376:1421–1426.

Morrison, J.L., and S.R. Humphrey. 2001. Conservation value of private lands for crested caracaras in Florida. Conservation Biology 15:675–684.

Murcia, C. 1995. Edge effects in fragmented forests: Implications for conservation. Trends in Ecology and Evolution 10:58–62.

Murcia, C., J. Aronson, H.H. Kattan, D. Moreno-Mateos, K. Dixon, and D. Simberloff. 2014. A critique of the ‘novel ecosystem’ concept. Trends in Ecology and Evolution 20:548–553.

Mutch, R.W. 1970. Wildland fire and ecosystems – A hypothesis. Ecology 51:1046–1051.

Myers, N., R.A. Mittermeier, C.G. Mittermeier, G.A.B. Da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature 403:853–858.

Nijhuis, M. 2021. Beloved Beasts: Fighting for Life in an Age of Extinction. WW. Norton and Company, New York, NY.

Nitzu, E., M. Vlaicu, A. Giurginca, I.N. Meleg, I. Popa, A. Nae, and S. Baba. 2018. Assessing preservation priorities of caves and karst areas using the frequency of endemic cave-dwelling species. International Journal of Speleology 47:43–52.

Nock, C.A., R.J. Vogt, and B.E. Beisner. 2016. Functional traits. Encyclopedia of Life Sciences. John Wiley & Sons, Ltd., Chichester, UK. doi:10.1002/9780470015902.a0026282

North, M.P., S.L. Stephens, B.M. Collins, J.K. Agee, G. Aplet, J.F. Franklin, and P.Z. Fulé. 2015. Reform forest fire management: Agency incentives undermine policy effectiveness. Science 349:1280–1281.

Noss, R.F. 1983. A regional landscape approach to maintain diversity. BioScience 33:700–706.

Noss, R.F. 1987a. From plant communities to landscapes in conservation inventories: A look at The Nature Conservancy (USA). Biological Conservation 41:11–37.

Noss, R.F. 1987b. Protecting natural areas in fragmented landscapes. Natural Areas Journal 7(1):2–13.

Noss, R.F. 1991. From endangered species to biodiversity. Pp. 227–246 in K. Kohm, ed. Balancing on the Brink of Extinction: The Endangered Species Act and Lessons for the Future. Island Press, Washington, DC.

Noss, R.F. 1995. Maintaining Ecological Integrity in Representative Reserve Networks. World Wildlife Fund Canada, Toronto, ON.

Noss, R.F. 1996. Ecosystems as conservation targets. Trends in Ecology and Evolution

11:351.

Noss, R.F. 2001. Beyond Kyoto: Forest management in a time of rapid climate change. Conservation Biology 15:578–590.

Noss, R.F. 2013. Forgotten Grasslands of the South: Natural History and Conservation. Island Press, Washington, DC.

Noss, R.F. 2018. Fire Ecology of Florida and the Southeastern Coastal Plain. University Press of Florida, Gainesville.

Noss, R.F., J.M. Cartwright, D. Estes, T. Witsell, G. Elliott, D. Adams, M. Albrecht, R. Boyles, P. Comer, C. Doffitt, et al. 2021. Improving species status assessments under the U.S. Endangered Species Act and implications for multispecies conservation challenges worldwide. Conservation Biology 35:1715–1724.

Noss, R.F., and A. Cooperrider. 1994. Saving Nature's Legacy: Protecting and Restoring Biodiversity. Island Press, Washington, DC.

Noss, R.F., E. Dinerstein, B. Gilbert, M. Gilpin, B. Miller, J. Terborgh, and S. Trombulak. 1999. Core areas: Where nature reigns. Pp. 99–128 in M.E. Soulé and J. Terborgh, eds., Continental Conservation: Scientific Foundations of Regional Reserve Networks. Island Press, Washington, DC.

Noss, R.F., J.F. Franklin, W.L. Baker, T. Schoennagel, and P.B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

Noss, R.F., and L.D. Harris. 1986. Nodes, networks, and MUM's: Preserving diversity at all scales. Environmental Management 10:299–309.

Noss, R.F., E.T. LaRoe, and J.M. Scott. 1995. Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation. Biological Report 28, USDI National Biological Service, Washington, DC.

Oh, B., K.J. Lee, C. Zaslawski, A. Yeung, D. Rosenthal, L. Larkey, and M. Back. 2017. Health and well-being benefits of spending time in forests: Systematic review. Environmental Health and Preventive Medicine 22:71.

Parks Canada. 2017. Guiding principles and operational policies. 2.0 Management planning. Parks Canada, Ottawa, ON.

Parrish, J.D., D.P. Braun, and R.S. Unnasch. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. BioScience 53:851–860.

Pearson, A.M. 2017. Force of Nature: George Fell, Founder of the Natural Areas Movement. University of Wisconsin Press, Madison.

Peters, R.L., and J.D.S. Darling. 1985. The greenhouse effect and nature reserves. BioScience 35:707–717.

Peterson St-Laurent, G., L.E. Oakes, M. Cross, and S. Hagerman. 2021. R–R–T (resistance–resilience–transformation) typology reveals differential conservation approaches across ecosystems and time. Communications Biology 4:39.

Phillips, A. 2004. The history of the international system of protected area management categories. Parks 14:4–14.

Pimm, S.L., C.N. Jenkins, R. Abell, T.M. Brooks, J.L. Gittleman, L.N. Joppa, P.H. Raven, C.M. Roberts, and J.O. Sexton. 2014. The biodiversity of species and their rates of extinction, distribution, and protection. Science 344(6187):1246752.

Pimm, S.L., C.N. Jenkins, and B.V. Li. 2018. How to protect half of Earth to ensure it protects sufficient biodiversity. Science Advances 4:eaat2616.

Poiani, K.A., B.D. Richter, M.G. Anderson, and H.E. Richter. 2000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. BioScience 50:133–146.

Preston, F.W. 1948. The commonness, and rarity, of species. Ecology 29:254–283.

Prober, S.M., V.A. Doerr, L.M. Broadhurst, K.J. Williams, and F. Dickson. 2019. Shifting the conservation paradigm: A synthesis of options for renovating nature under climate change. Ecological Monographs 89:e01333.

Rabinowitz, D., S. Cairns, and T. Dillon. 1986. Seven forms of rarity and their frequency in the flora of the British Isles. Pp. 182–204 in M.E. Soulé, ed., Conservation Biology: The Science of Scarcity and Diversity. Sinauer, Sunderland, MA.

Reed, S.E., and A. Merenlender. 2008. Quiet, nonconsumptive recreation reduces protected area effectiveness. Conservation Letters 1:146–154.

Reid, A.M., L. Morin, P.O. Downey, L. French, and J.G. Virtue. 2009. Does invasive plant management aid the restoration of natural ecosystems? Biological Conservation 142:2342–2349.

Richardson, D.M., J.J. Hellmann, J. McLachlan, D.F. Sax, M.W. Schwartz, J. Brennan, P. Gonzalez, T. Root, O. Sala, S.H. Schneider, et al. 2009. Multidimensional evaluation of managed relocation. Proceedings of the National Academy of Sciences 106:9721–9724.

Riley, J.L., and P. Mohr. The natural heritage of southern Ontario's settled landscapes. Technical Report TR-001, Ontario Ministry of Natural Resources, Aurora.

Rohwer, Y., and E. Marris. 2021. Ecosystem integrity is neither real nor valuable. Conservation Science and Practice 3(4):3411.

Romps, D.M., J.T. Seeley, D. Vollaro, and J. Molinari. 2014. Projected increase in lightning strikes in the United States due to global warming. Science 346:851–854.

Rowe, J.S. 1976. The significance of natural areas. Proceedings of a Symposium of the Thirteenth Annual Meeting of the Canadian Botanical Association.

Rowe, J.S. 1990. Home Place: Essays on Ecology. NeWest Publishers, Edmonton, AB.

Ryan, K.C., E.E. Knapp, and J.M. Varner. 2013. Prescribed fire in North American forests and woodlands: History, current practice, and challenges. Frontiers in Ecology and the Environment 11:e15–e24.

Safford, H.D, and J.T. Stevens. 2017. Natural range of variation (NRV) for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report PSW-GTR-256, USDA Forest Service, Pacific Southwest Research Station, Albany, CA.

Safford, H.D., J.A. Wiens, and G. Hayward. 2012. The growing importance of the past in managing ecosystems of the future. Pp. 319–327 in J.A. Wiens, G. Hayward, H.D. Safford, and C.M. Giffen, eds., Historical Environmental Variation in Conservation and Natural Resource Management. John Wiley and Sons, New York, NY.

Safford, H.D., J.W. Wright, and Region 5 and PSW Research Station RNA Committee. 2015. Suggested fire suppression and rehabilitation policy for Research Natural Areas (RNAs) in Region 5. Version 2. https://www.fs.fed.us/psw/rna/index.shtml

Schoennagel, T., T.T. Veblen, and W.H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661–676.

Schwartz, M.W., J.J. Hellmann, J.M. McLachlan, D.F. Sax, J.O. Borevitz, J. Brennan, A.E. Camacho, G. Ceballos, J.R. Clark, H. Doremus, et al. 2012. Managed relocation: Integrating the scientific, regulatory, and ethical challenges. BioScience 62:732–743.

Schwegman, J. 1981. Letter from the President. Journal of the Natural Areas Association 1(4):2.

Scott, J.M., F. Davis, B. Csuti, R. Noss, B. Butterfield, C. Groves, H. Anderson, S. Caicco, F. D'Erchia, T.C. Edwards, et al. 1993. Gap Analysis: A geographic approach to protection of biological diversity. Wildlife Monographs 123:1–41.

Shanklin, J.F. 1968. Society affairs: Natural areas project. Journal of Forestry 66:873–879.

Sheley, R.L., and J. Krueger-Mangold. 2003. Principles for restoring invasive plant-infested rangeland. Weed Science 51:260–265.

Shelford, V.E., ed. 1926. Naturalist's Guide to the Americas. Committee on the Preservation of Natural Conditions of the Ecological Society of America. Williams and Wilkins, Baltimore, MD.

Simberloff, D., and J. Cox. 1987. Consequences and costs of conservation corridors. Conservation Biology 1:63–71.

Soulé, M.E., ed. 1987. Viable Populations for Conservation. Cambridge University Press, Cambridge, UK.

Soulé, M.E., D.T. Bolger, A.C. Alberts, J. Wright, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation Biology 2:75–92.

Soulé, M.E., J.A. Estes, J. Berger, and C. Martinez del Rio. 2003. Ecological effectiveness: Conservation goals for interactive species. Conservation Biology 17:1238–1250.

Soulé, M.E., J.A. Estes, B. Miller, and D.L. Honnold. 2005. Strongly interacting species: Conservation policy, management, and ethics. BioScience 55:168–176.

Soulé, M.E., and B.A. Wilcox, eds. 1980. Conservation Biology: An Evolutionary-Ecological Perspective. Sinauer, Sunderland, MA.

Stein, B.A., L.S. Kutner, and J.S. Adams, eds. 2000. Precious Heritage: The Status of Biodiversity in the United States. Oxford University Press, New York, NY.

Stein, B.A., A. Staudt, M.S. Cross, N.S. Dubois, C. Enquist, R. Griffis, L.J. Hansen, J.J. Hellmann, J.J. Lawler, E.J. Nelson, and A. Pairis. 2013. Preparing for and managing change: Climate adaptation for biodiversity and ecosystems. Frontiers in Ecology and the Environment

11:502–510.

Swengel, A.B., and S.R. Swengel. 2007. Benefit of permanent non-fire refugia for Lepidoptera conservation in fire-managed sites. Journal of Insect Conservation 11:263–279.

Taschereau, P. 1984. The Canadian approach to natural areas protection. Natural Areas Journal 4(1):4–10.

Terborgh, J., and B. Winter. 1980. Some causes of extinction. Pp. 119–133 in M.E. Soulé and B.A. Wilcox, eds., Conservation Biology: An Evolutionary-Ecological Perspective. Sinauer, Sunderland, MA.

Tierney, G.L., D. Faber-Langendoen, B.R. Mitchell, W.G. Shriver, and J.P. Gibbs. 2009. Monitoring and evaluating the ecological integrity of forest ecosystems. Frontiers in Ecology and the Environment 7:308–316.

Unnasch, R.S., D.P. Braun, P.J. Comer, and G.E. Eckert. 2018. The ecological integrity assessment framework: A framework for assessing the ecological integrity of biological and ecological resources of the national park system (Version 1.1). Natural Resource Report NPS/NRSS/BRD/NRR—2018/1602, National Park Service, Fort Collins, CO.

[USFS] U.S. Forest Service. 1966. Forest Service Manual. USDA Forest Service, Washington, DC.

[USGCRP] U.S. Global Change Research Program. 2017. Climate science special report—Fourth National Climate Assessment, v. I. D. Wuebbles, D. Fahey, K. Hibbard, D. Dokken, B. Stewart, and T. Maycock, eds. U.S. Global Change Research Program, Washington, DC.

[USGCRP] U.S. Global Change Research Program. 2018. Climate science special report—Fourth National Climate Assessment, v. II. U.S. Global Change Research Program, Washington, DC.

Van Wagtendonk, J.W., ed. 2018. Fire in California's Ecosystems. University of California Press, Berkeley.

Veldman, J.W., J.C. Aleman, S.T. Alvarado, T.M. Anderson, S. Archibald, W.J. Bond, T.W. Boutton, N. Buchmann, E. Buisson, J.G. Canadell, et al. 2019. Comment on "The global tree restoration potential." Science 366(6463):10.1126/science.aay7976.

Veldman, J.W., G.E. Overbeck, D. Negreiros, G. Mahy, S. Le Stradic, G.W. Fernandes, G. Durigan, E. Buisson, F.E. Putz, and W.J. Bond. 2015. Where tree planting and forest expansion are bad for biodiversity and ecosystem services. BioScience 65:1011–1018.

Waldrop, T.A., and S.L. Goodrick. 2012. Introduction to Prescribed Fire in Southern Ecosystems. USDA Forest Service, Southern Research Station, Asheville, NC.

Walston, L.J., and H.M. Hartmann. 2018. Development of a landscape integrity model framework to support regional conservation planning. PLOS One 13(4):e0195115.

Watson, J.E.M., N. Dudley, D.B. Segan, and M. Hockings. 2014. The performance and potential of protected areas. Nature 515:67–73.

West, J.M., S.H. Julius, P. Kareiva, C. Enquist, J.J. Lawler, B. Petersen, A.E. Johnson, and M.R. Shaw. 2009. U.S. natural resources and climate change: Concepts and approaches for management adaptation. Environmental Management 44:1001–1021.

Wilcove, D.S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. BioScience 48:607–615.

Wilhelm, G. 1977. Ecological Assessment of Open Land Areas in Kane County, Illinois: A Checklist of the Kane County Flora With Numerical Evaluations – Its Basis, Rationale and Application. Kane County Urban Development, Geneva, IL.

Williams, B.K. 2011. Adaptive management of natural resources – Framework and issues. Journal of Environmental Management 92:1346–1353.

Woodley, S., J. Kay, and G. Francis. 1993. Ecological Integrity and the Management of Ecosystems. St. Lucie Press, Boca Raton, FL.

Wulf, A. 2015. The Invention of Nature: Alexander von Humboldt's New World. Knopf, New York, NY.