

Technical Reference  
On  
Using Surrogate Species for  
Landscape Conservation  
US Fish & Wildlife Service



Photo by Ken Greshowak



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**On Using Surrogate Species for Landscape Conservation**  
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## 1 INTRODUCTION

Efforts to identify important conservation areas and their associated ecological processes and features can face substantial taxonomic, logistical, and financial challenges. To address these challenges, many agencies and conservation organizations rely on surrogate species that are intended to serve as proxies for large suites of species, biotic communities, habitats, and ecosystems. The term “surrogate species” encompasses a wide variety of terms: umbrella species, focal species, landscape species, management indicators, environmental indicators, biodiversity indicators, keystone species, flagship species and environmental surrogates.

This reference document provides a common understanding of the concepts, terms and definitions associated with surrogate species. Published and unpublished literature are summarized to cover how surrogate species are defined, identified and selected, and used, along with critiques of their value as management tools.

Finally, this document reviews the effectiveness of achieving desired conservation outcomes via different surrogate species approaches. Although empirical reviews of surrogate species approaches are limited, they suggest that some surrogate approaches can facilitate achieving a conservation outcome, particularly when conservation goals and objectives are clearly defined at the outset and appropriate criteria are used to select the surrogate species. This reference document concludes with a discussion of the various monitoring and evaluation methodologies that are available to assess the effectiveness of surrogate species.

### 1.1 LAYOUT OF THIS DOCUMENT

**Section 2** reviews the literature supporting the concept of surrogacy and the various terms that are subsumed under the word “surrogate.” **Section 3** reviews the literature on how surrogates and related terms have been used to advance species, habitat, and ecosystems conservation. **Section 4** reviews the literature on selecting surrogate species. **Section 5** reviews the literature on the validity of the surrogacy assumption that underlies the use of all surrogate species. **Section 6** reviews the literature on the efficacy of surrogate species and the usefulness of surrogate species as a tool for implementing a conservation plan or program.

## 2 RELEVANT TERMS

### 2.1 INTRODUCTION

Surrogate approaches are used often and for many reasons in conservation planning (Wiens et al. 2008; Caro 2010; Brock & Atkinson 2013). When cost or system complexities make it expensive or impossible to directly assess every species or component of an ecosystem, the use of surrogates may be warranted. Generally, surrogates can be categorized as abiotic and biotic. Abiotic surrogates often consist of climate, geological, soils, water quality, water quantity, and other physical parameters of an ecosystem (Caro 2010). Biotic surrogates can range from vegetative communities and various community measures to specific species or habitat characteristics.

This document focuses on the use of surrogate species and their relationship to conservation planning. The word “surrogate species” encompasses a wide variety of terms, many of which are used in ways that are fundamentally different from their original meaning. Wiens et al. (2008) and Caro (2010) define “surrogate species” as species used to represent other species<sup>1</sup> (or, less commonly, aspects of the environment) to attain a conservation objective. The scientific literature regarding the definition and use of surrogate species in conservation planning is exhaustive (e.g., Landres et al. 1988; Martino et al. 2005; Favreau et al. 2006; Rodrigues & Brooks 2007; Wiens et al. 2008; Caro 2010; Mellin et al. 2011; Brock & Atkinson 2013). Unfortunately, as reviewed by a number of publications, there is much confusion and misuse of surrogate species terms, even within the scientific literature (Caro et al. 2005; Caro 2010; Heink & Kowarik 2010a; Veríssimo et al. 2011; Isasi-Catala 2011; Barua 2011).

There are several terms that refer to the co-occurring species that are likely to benefit from conservation activities directed at a surrogate species. *Background species* is commonly used in relationship to surrogate species (Caro 2010), but also occasionally used with other meanings. *Target species* has been frequently used for this purpose (Caro 2010), but has also been used in other contexts in conservation planning. *Beneficiary species* most descriptively identifies the relationship of the species to the surrogate species (after Roberge & Angelstam 2004) and refers to the co-occurring species that may benefit from conservation activities directed at the surrogate species.

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<sup>1</sup> For the purposes of this document, we use the term *beneficiary species* to refer to the co-occurring species that are likely to benefit from conservation activities directed at a surrogate species (after Roberge & Angelstam 2004).

The FWS has a long history of using surrogate species beginning with the concept of “evaluation species.” “Evaluation species” were a feature of the FWS’ Habitat Evaluation Procedures (HEP) to “quantify habitat suitability and determine changes in the number of available” habitat units (FWS 1976, 1980). A critical assumption of HEP was that impacts to evaluation species could be extrapolated to species that were part of a larger biotic community if the evaluation species were selected carefully. According to the HEP procedures, species selected as “evaluation species” should (a) have known sensitivities to specific land use actions (to serve as “early warning indicators”); (b) perform a key role in a community because of their role in nutrient cycling or energy flows; or (c) represent groups of species that use a common environmental resource. These groups were referred to as “guilds,” representative species were selected from each guild, and the response of the “evaluation species” to environmental change were assumed to be representative of the other members of the “guild.”

In 1981, the FWS codified the term “evaluation species” in its mitigation policy (46 Federal Register 7662). That policy defined an “evaluation species” to mean “those fish and wildlife resources in the planning area that are selected for impact analysis. They must currently be present or known to occur in the planning area during at least one stage of their life history.” The policy created two exceptions to this practice: (1) where species not present in a planning area have been identified in fish and wildlife restoration or improvement plans approved by state or federal resource agencies, or (2) where a species’ occurrence in a planning area will result from natural species’ succession over the life of the project.

“Evaluation species” represented species with high public interest, economic value, or both or they provided a broader ecological perspective of an area. Adopting language from the HEP procedures, the second of these categories included (1) species “known to be sensitive to specific land and water use actions. The species selected with this approach serve as ‘early warning’ or indicator species for the affected fish and wildlife community”; (2) species that “perform a key role in a community because of their role in nutrient cycling or energy flows”; and (3) species that “represent groups of species which utilize a common environmental resource (guilds)” (46 Federal Register 7662).

Further evolution of the use of surrogate species by the FWS occurred in 2006, when the agency adopted Strategic Habitat Conservation as a tool for working through partnerships to develop and implement an iterative approach to landscape conservation. Because working at landscape or larger geographic scales makes it impractical to consider the needs of all species, the *Strategic Habitat Conservation Final Report of the National Ecological Assessment Team* (FWS & USGS 2006) introduces the use of focal species as surrogate species for conservation planning and developing conservation designs. The *Strategic Habitat Conservation Handbook* (FWS 2008) also discusses selecting “a subset of focal

species to represent the needs of larger guilds of priority species that use habitats and respond to management similarly.”

Most recently, as part of implementing SHC, the FWS has been exploring and testing the use of surrogate species for designing landscapes; this reference document is one tool that has been developed to help the FWS and its conservation partners determine whether it is sensible to use surrogate species as tools for conservation planning, landscape design and implementation efforts at the landscape or larger geographic scales,

Because a common understanding of concepts, terms, and definitions is a pre-requisite for effective management action, the remainder of this document discusses the various terms that are included under the general characterization of “surrogate species” and summarizes the diversity of their usage. Each narrative describes the different terms, to the extent that they have been defined, with supporting information that helps convey the breadth of each term’s usage.

## 2.2 UMBRELLA SPECIES

The term *umbrella species* has been used to pursue so many different conservation objectives that there is little consensus on a specific definition (Roberge & Angelstam 2004; Caro 2010). Despite the absence of a common definition, most uses of the concept have assumed that the presence of a particular species (the umbrella species) in a geographical area indicates that other species (the beneficiary species) will also be present, so focusing conservation effort on the umbrella species will benefit other members of the biotic community (Berger 1997; Zacharias & Roff 2001; Roberge & Angelstam 2004; Favreau et al. 2006; Caro 2010).

Three types of umbrella species have been recognized (Zacharias & Roff 2001; Caro 2010):

- Classic umbrella species – In the classic application, the location, size, and configuration of areas occupied by viable populations of one species (the umbrella species) is assumed to identify sites or areas in larger landscapes or regions that support viable populations of beneficiary species (Berger 1997; Zacharias & Roff 2001; Caro 2003, 2010).
- Local umbrella species – With local umbrella species, the occurrence of one species (the umbrella species) in a particular geographic area is assumed to “indicate” or predict the occurrence of beneficiary species, so reserves that are delineated to meet the habitat requirements of the umbrella species will contain as many of the beneficiary species as possible (Caro 2010). This concept corresponds with the “site-selection umbrella species” concept discussed by Roberge & Angelstam (2004).
- Management umbrella species – This type of surrogate species combines criteria distinguishing management and environmental indicator species and emphasizes



one species serving as a general surrogate for other species (Zacharias & Roff 2001; Roberge & Angelstam 2004). Management umbrella species “may predict population responses (of other species) to planned human activities” (Caro 2010).

In all three cases, umbrella species selected have typically been long-lived habitat generalists that required large areas to maintain minimum viable populations (Caro & O’Doherty 1999; Fleishman et al. 2001). Early applications of the umbrella species concept focused on large mammals and birds, but latter applications extended the concept to invertebrates (Roberge & Angelstam 2004). In most cases, umbrella species have been used as a short-cut to overcome the challenge of assessing the abundance and viability of populations of many different species that occur in a region, landscape, reserve, or management unit.

Because umbrella species have been used to “indicate” the presence of beneficiary species, there is conceptual overlap between umbrella species and “biodiversity indicators” (discussed in **Section 2.5.3**). However, the two concepts have been used at different spatial scales: umbrella species have typically been used to identify conservation areas at regional or local spatial scales while biodiversity indicators have been used at global or continental levels (Caro 2010). Unfortunately, the terms are often used interchangeably at the regional level (Caro 2010). See **Section 3.2** for examples and more on the methodology. See **Section 4.2** for criteria used to select umbrella species. See **Section 5.2.1** for a discussion of the validity of the assumptions. See **Section 6.1** for a review of the efficacy of their use.

## 2.3 FOCAL SPECIES

After arguing that a single species is not likely to represent the diversity of habitat needs of all of the species that occur in ecosystems or landscapes, Lambeck (1997) argued that it would be more appropriate to use a set of species rather than a single umbrella species. He recommended using a suite of species, each of which is used to define the characteristics of different landscape attributes that must be represented in the landscape. He called this set of species “focal species” and argued that the spatial, compositional, and functional requirements of these “focal species” could be used to develop explicit guidelines regarding the composition, quantity, and configuration of habitat patches and the management regimes that must be applied to the resulting design of the landscape.

Caro (2010) argued that this concept of “focal species” was reasonably precise, could work well, and provided examples from Australia and Nova Scotia (Watson et al. 2001) to support his argument. However, he also noted a variety of situations in which focal species do not work and pointed to challenges that make identifying these species difficult (including poor overall data quality, limited numbers of species for which data are available, etc.). His larger complaint was that many authors had used the term “focal

species” to refer to different concepts, eliminating the precision of Lambeck’s (1997) original definition (see **Table 1**).



**Figure 2. Plains Bison (*Bison bison*) is a focal species for North American prairie landscapes. It meets the criteria for a focal species (Lambeck 1997) in that it encompasses some of the most demanding requirements of the functional landscape and thus will encompass the requirements of other species.**

**Table 1. Examples of the Diversity of Focal Species Definitions**

Focal Species Definition	Examples
“suite of species, each of which is used to define different spatial and compositional attributes that must be present in a landscape and their appropriate management regimes”	Lambeck 1997
Focal species is a term that includes multiple categories, including, but not limited to, indicators, keystones, umbrellas, special interest species, and flagships.	Dale & Beyeler 2001; Kautz & Cox 2001; Zacharias & Roff 2001; Noss et al. 2002
Focal species as a term referring to the landscape species selected for use in landscape design.	Brock & Atkinson 2013
Focal species as a term referring to the species being studied.	Rubino & Hess 2003; Basset et al. 2008
Focal species as a term referring to surrogate species generally.	FWS 2008; Caro & O’Doherty 1999
*See the remainder of this section for definitions for each term and <b>Table 2</b> for a summary of how they are used.	

The *Strategic Habitat Conservation Final Report* of the National Ecological Assessment Team and the *Strategic Habitat Conservation Handbook: A Guide to Implementing the Technical Elements of Strategic Habitat Conservation (Version 1.0)* illustrate this complaint (FWS & USGS 2006; FWS 2008). The SHC Report uses the term “focal species” in a way similar to Lambeck (1997) while the SHC Handbook uses the term “focal species” to refer to any surrogate approach as well as some non-surrogate approaches. For clarity of purpose, when the term focal species is used, it should be used as Lambeck (1997) originally defined it: as a suite of species whose spatial, compositional, and functional requirements are representative of the needs of a larger pool of species (also see discussion of Butler et al. (2012) in **Section 4.3** for discussion of criteria for selecting “focal species”).

## 2.4 LANDSCAPE SPECIES

Sanderson et al (2002) defines *landscape species* as those that use large, diverse areas (either species with large home ranges or species with wide distributions) and often have significant impacts on the structure and function of natural ecosystems. A landscape species integrates elements from several types of surrogate species and results in the ability to define the conservation needs of a landscape by both the extent of the area and the important variation within it, including ecological processes and considering anthropogenic threats (Sanderson et al. 2002; Brock & Atkinson 2013). Studies using landscape species are generally more quantitative than studies using historical umbrella species (Caro 2010; Brock & Atkinson 2013). See **Section 3.2.2** for examples and more on the methodology. See **Section 4.2** for the criteria used to identify landscape species. See **Section 5.2.3** for a discussion of the validity of the assumptions. See **Section 6.1** for a review of the efficacy of their use.

## 2.5 INDICATOR SPECIES

For more than a century, investigators have used species as indicators of environmental conditions, indicators of management activity, or indicators of biodiversity. For example, Clements (1916) and Shantz (1911) used plant species as indicators of lands that might be suitable for grazing or agriculture while Bobrov (1955) and Middleton (1956) used plant species as indicators of air quality.

Moore (1962) is reported to have been the first to explicitly use “indicator species” as a conservation tool, although he did not define the term (Favreau et al. 2006). Landres et al. (1988) defined an indicator species as “an organism whose characteristics (e.g., presence or absence, population density, dispersion, reproductive success) are used as an index of attributes too difficult, inconvenient, or expensive to measure for other species or environmental conditions of interest.”

Heink & Kowarik (2010a) defined the term broadly: “an indicator in ecology and environmental planning is a component or a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals. Environmentally relevant phenomena are pressures, states, and responses as defined by the [Organisation for Economic Co-Operation Development] (OECD 2003).” The narratives that follow discuss the three main categories of indicator species that have appeared in literature: environmental conditions indicators, management indicators, and biodiversity indicators.

### 2.5.1 ENVIRONMENTAL CONDITIONS INDICATORS

*Environmental conditions indicators* are commonly used as surrogates to monitor the effects of environmental conditions on ecological systems, including the species supported by those conditions (Dale & Beyeler 2001; Lindenmayer & Likens 2011). Species selected and monitored as environmental indicators are sensitive to particular environmental conditions and are considered representative of other species that require the same or similar environmental conditions. As a result, changes in populations of indicator species are assumed to be representative of changes in beneficiary species (e.g., deteriorating water quality reduces a population of indicator fish species and is assumed to affect other species in that same ecosystem). Many different terms are used to describe environmental conditions indicators, including, but not limited to, disturbance indicator, environmental indicator, cross-taxon indicator, climate change indicator and predation indicator. See **Section 3.3** for examples and more on the methodology. See **Section 4.3** for the criteria used to select environmental conditions indicators. See **Section 5.3** for a discussion of the validity of the assumptions. See **Section 6.3** for a review of the efficacy of their use.

### 2.5.2 MANAGEMENT INDICATORS

Species have been selected as *management indicators* to evaluate and support land and conservation management decisions for more than a century and often include species that are indicators of environmental conditions (see above). As discussed in the introductory paragraph to Section 2.5, Shantz (1911) and Clements (1916) used plant communities as indicators to assess land for potential grazing and agricultural uses. Species have also been used to monitor air pollution (Bobrov 1955; Middleton 1956; Weinstein & McCune 1970), water pollution (Cairns, Jr et al. 1973; Cairns, Jr 1977), and other toxins for more than half a century. In a special supplement on the effects of pesticides on wildlife published in 1966, the British Ecological Society discussed the importance of using species to monitor contamination and presented criteria for selecting indicator species (Moore 1966).

Specifically, Landres (1988) described *management indicators* as species used to track the effects of management actions on the species potentially affected by the management. A few species are selected as indicators and monitored with the assumption that other species subject to the management action are similarly affected. The terms used to for management indicators include “management indicator” and “management umbrella” species.

The U.S. Forest Service (USFS) adopted the term “management indicator species” in 1982 in regulations the agency published to implement the National Forest Management Act (47 FR 43037). These regulations required “management indicator species” to be selected and used as a basis for “estimating the effects of each alternative on fish and wildlife populations” and for monitoring (36 CFR 219.19). The regulations further stated that “[t]hese species shall be selected because their population changes are believed to indicate the effects of management activities” and that “[f]ish and wildlife habitat shall be managed to maintain viable populations of existing and native and desired non-native vertebrate species in the planning area.” Except for threatened and endangered species, the regulations did not require forest plans to establish a specific population level for management indicator species; rather, the management indicator species was treated as a proxy measure of the effects of management on the overall welfare (population viability) of beneficiary species.

In 1991 amendments to the Forest Service Manual, the USFS defined the term *management indicators* as “plant and animal species, communities, or special habitats selected for emphasis in planning, and which are monitored during forest plan implementation in order to assess the effects of management activities on their populations and the populations of other species with similar habitat needs which they may represent” (USFS 1991). Those amendments also defined the term ecological indicators to mean “plant or animal species, communities, or special habitats with a narrow range of ecological tolerance. Such indicators are selected for emphasis and monitored during forest plan implementation because their presence and relative abundance serve as a barometer of ecological conditions within a management unit.” See **Section 3.3** for examples and more on the methodology. See **Section 4.3** for the criteria used to select environmental conditions indicators. See **Section 5.3** for a discussion of the validity of the assumptions. See **Section 6.3** for a review of the efficacy of their use<sup>2</sup>.

### 2.5.3 BIODIVERSITY INDICATORS

Biodiversity or “biological diversity” generally refers to the variety and variability of living organisms and the environments in which they occur; however, it also encompasses concepts such as ecological health, sustainability, and resilience (US Congress, Office of Technology Assessment 1987; Heywood 1995; Caro 2010). From a management perspective, the goal of conserving biodiversity is to prevent biotic impoverishment and to counter its causes (Scott et al. 1993). *Biodiversity indicators* defined as surrogates are intended to

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<sup>2</sup> Also see Schultz, C.A., T.D. Sisk, B.R. Noon, and M.A. Nie. 2013. Wildlife conservation planning under the United States Forest Service’s 2012 planning rule. *Journal of Wildlife Management* 77:428–444.

convey meaningful information about something larger than the surrogate itself (Duelli & Obrist 2003; Biodiversity Indicators Partnership 2011). Indicators are typically well-understood (taxonomically and ecologically), easily monitored, occur in various environmental conditions, and show strong relationships with beneficiary species, communities, ecosystems, or other phenomena (Noss 1990; Pearson 1994; Duelli & Obrist 2003). Depending on their use, biodiversity indicators can be used to identify areas that are important for conservation or function as environmental or management indicators. See **Section 3.2.1** for more discussion on the various uses of biodiversity indicators. See **Section 4.4** for the criteria used to select biodiversity indicators. See **Section 5.2.4** for more discussion on the validity of assuming one taxa can serve as a biodiversity indicator for other taxa. See **Section 6.2** for a discussion of the efficacy of their use.

## 2.6 FLAGSHIP SPECIES

Caro and O'Doherty (1999) define *flagship species* as those species that are used to reflect or engender public support for conservation efforts. Veríssimo et al. (2011) defined flagship species as “a species used as the focus of a broader conservation marketing campaign based on its possession of one or more traits that appeal to the target audience.” As summarized in Barua et al. (2011), there are many uses of flagship species, including

- Conservation awareness
- Fund raising
- Promoting ecotourism
- Community-based conservation
- Promotion of funded research
- Protection of species/habitat
- Influencing policy

Historically, flagship species were rarely identified or verified as having a positive effect on beneficiary species, other than by how much fundraising was accomplished and, therefore, did not truly serve as a surrogate species (Barua et al. 2011; Veríssimo et al. 2011). Recent advances in understanding how a flagship species can be used to benefit other species have identified more specific ways of selecting flagship species and verifying their effectiveness, resulting in greater surrogacy effectiveness (Barua et al. 2011; Veríssimo et al. 2011). During an extensive literature review on flagship species, Veríssimo et al. (2011) state that flagship species should be selected for their marketing value for achieving a specific conservation objective. A flagship performs as a surrogate when improved public awareness of the flagship species results in greater appreciation for the landscape *and* improved conservation outcomes of both the flagship and other species (Veríssimo et al. 2011).

See **Section 3.4** for examples and more on the methodology. See **Section 4.5** for the criteria used to select flagship species. See **Section 5.4** for a discussion of the validity of the assumptions. See **Section 6.4** for a review of the efficacy of their use.

## 2.7 KEYSTONE SPECIES

The term keystone species was introduced by Paine (1969) who defined it as a ‘species whose population is “the keystone of the community’s structure,” whereby the integrity and stability of the community are determined by its activities and abundance.’ Although Paine (1969, 1995) initially referred to species that maintained the stability of biotic communities and ecosystems, that focus was lost in subsequent development of the term (Cottee-Jones & Whittaker 2012).

Power et al. (1996) defined a *keystone species* as a species “whose impact on its community is large, and disproportionately large relative to its abundance.” An alternate definition of a keystone species is provided by Davic (2003) where “a keystone species is a strongly interacting species whose top-down effect on species diversity and competition is large relative to its biomass dominance within a functional group.” Darwall & Vié (2005) extended Paine’s original definition further by defining “keystone species” as “a species whose loss from an ecosystem would cause a greater than average change in other species populations or ecosystem processes; whose continued wellbeing is vital for the functioning of a whole community.”

Many authors refer to keystone species as surrogates when describing their studies or conservation methodologies (Simberloff 1998; Linnell et al. 2000; Zacharias & Roff 2001; Martino et al. 2005; Favreau et al. 2006; Caro 2010). However, keystone species are defined by their role in biotic communities or ecosystems (Mills et al. 1993; Cottee-Jones & Whittaker 2012) and would not be “true” surrogate species unless they represent other species or aspects of the environment (Davic 2003; Isasi-Catala 2011). Favreau et al. (2006) found no studies that tested the assumption that a keystone species was functioning as a surrogate species.

Caro (2010) summarized a number of problems with using keystone species during conservation planning, including their highly variable effects on the integrity and stability of communities and ecosystems, both in time and location. Despite the importance of their ecological roles — or perhaps because of it — “keystone” species generally are not proxies for other species. **Selecting species because of their role in maintaining the ecology of a community or system is not the same as selecting species because they are representative of other species.** Before they could be used as surrogates, conservation efforts would need to ensure that “keystone” species represent other species or aspects of the environment.

## 2.8 ENVIRONMENTAL SURROGATES

Some surrogate concepts do not use species as the surrogates. Instead, these concepts, often referred to as *environmental surrogates*, use measurements of habitat or other direct physical measurements as surrogates for conservation targets<sup>3</sup> (Ferrier & Watson 1997; Caro 2010; Lindenmayer & Likens 2011). Various habitat and direct measurements may be used in conjunction with surrogate species during a comprehensive planning effort or design processes. See **Section 3.2.4** for more discussion about the use of environmental surrogates for selecting conservation areas. See **Section 5.2.5** for a discussion of the validity of the surrogacy assumption for environmental surrogates.

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<sup>3</sup> Conservation target - An element of biodiversity at a project site, which can be a species, habitat, or ecological system that a project has chosen to focus on. All targets at a site should collectively represent the species of conservation interest at the site. Adapted from Conservation Measures Partnership [www.ConservationMeasures.org](http://www.ConservationMeasures.org).



## 3 USES OF SURROGATE SPECIES

### 3.1 INTRODUCTION

The ways in which the different terms subsumed under the word “surrogate” have been used is diverse. For example, the types of surrogates that have been used for conservation efforts range from using physical measurements as proxies for ecosystem processes or suitable habitat to using one or more species as proxies for biotic communities, ecosystems, landscapes, and watersheds. (Landres et al. 1988; Zacharias & Roff 2001; Martino et al. 2005; Wiens et al. 2008; Fleishman & Murphy 2009; Caro 2010; Brock & Atkinson 2013). Nevertheless types of surrogate species have been generally grouped into three major categories of use (e.g., Caro 2010; Brock and Atkinson 2013):

1. Species used to define areas of conservation interest (**Section 3.2**)
  - biodiversity indicators
  - umbrella species
  - focal species
  - landscape species
2. Species used to document effects of environmental or management conditions (**Section 3.3**)
  - management indicator
  - environmental indicator
  - management umbrella species
  - cross-taxon indicator
3. Species used to engender public support (**Section 3.4**)
  - flagship species

*Identifying conservation goals and objectives is a necessary first step for selecting surrogates and leads to selection of the surrogate approach and the species selection criteria*  
(Caro 2010, Wiens et al. 2008).

Many authors have emphasized that when using surrogate species, conservation objectives and planning assumptions must be explicitly stated and subsequently monitored and tested (see **Section 5** and **6.5** for further discussion). This allows for an evaluation of the effects of the conservation actions on the surrogate and beneficiary species populations, as well as the effectiveness of the use of the surrogate species in achieving the conservation goal or objective (Caro 2010; Veríssimo et al. 2011; Murphy et al. 2011; Brock & Atkinson 2013; Schultz et al. 2013; Lindenmayer et al. 2014).

**Table 2. Summary of Surrogate Species Approaches and Related Terms**

Surrogate Species Term	Group Used	Beneficiary Species	Spatial Scale	Data Needed	Role in Landscape Conservation
<i>Conservation Objective: Define Conservation Areas (Examples in Section 3.2)</i>					
Biodiversity Indicator	Several taxa, biotic communities, ecosystems	All other taxa	Global, Continental, National, Regional	Highly variable depending on the component of diversity being measured	Conservation planning (Identify biodiversity hotspots and areas of biological significance)
Focal Species (sensu Lambeck 1997)	Populations of several species	Other species and populations	National, Regional	Population data, spatial requirements, resource limitations, habitat associations	Conservation planning (Identify limiting factors; identifying conservation areas)
Umbrella Species (Classic/Local)	One or few populations	Other taxa, all other taxa, other populations	Continental, National, Regional, Local	Population viability, habitat associations, spatial requirements, resource limitations	Conservation planning (Identify location and size of reserve(s) in general)
Landscape Species	A few species	Other species and populations	Continental, National, Regional, Local	Population viability, habitat associations, spatial requirements, sensitivity to disturbance, role in structure and function of landscapes	Landscape design (Identify location, size, configuration of conservation areas; more rigorous criteria than umbrella species; broader inclusion than just reserves)
<i>Conservation Objective: Effects of Environmental or Management Conditions (Examples in Section 3.3)</i>					
Environmental Indicator	One or few species	Other taxa, other species' and populations	Regional, Local	Natural history, ecological factors affecting them	Conservation planning (Assess extent of environmental change)
Umbrella Species (Management)	One or few populations	Other species' populations	National, Regional	Population data, information on management actions and species responses	Monitoring (Conservation target for management action and assess response to management, assumed to represent other species; generally less specific/ rigorous than management indicators)
Management Indicator	One or few species	Other species' populations	Regional, Local	Natural history, habitat associations, population levels	Monitoring (Assess management; may or may not be a conservation target)
Biodiversity Indicator	Several taxa, biotic communities, ecosystems	All other taxa	Global, Continental, National, Regional	Highly variable depending on the component of diversity being measured	Monitoring to assess the effects of management on ecosystems and biotic communities (ecological sustainability, health,

Table 2. Summary of Surrogate Species Approaches and Related Terms					
Surrogate Species Term	Group Used	Beneficiary Species	Spatial Scale	Data Needed	Role in Landscape Conservation
					and resilience)
<i>Conservation Objective: For Public Support</i> (Examples in Section 3.4)					
Flagship Species	One species	Habitat and all species within it	Global, Continental, National, Regional	Relevance to target audience, basic information about the flagship species	Conservation delivery (Build public support and/or raise funds)
<i>Sources:</i> Modified from Table 10-1 in Caro (2010) with additional detail from Brock & Atkinson (2013)					

### 3.2 USING SURROGATE SPECIES TO DEFINE IMPORTANT CONSERVATION AREAS

To avoid the taxonomic, logistical, and financial challenges that accompany efforts to identify important conservation areas by collecting and analyzing extensive data on multiple species, communities, and habitats, many efforts rely on surrogates. For example biodiversity assessments to identify areas that merit conservation efforts; landscape designs encompassing fish and wildlife habitat, important ecological values, functions and processes, and patterns of environmental change to inform conservation delivery across a landscape; and reserve selection to identify the location, size, and configuration of reserves – all rely on surrogates to overcome the challenges that would otherwise accompany these efforts. The surrogate species terms commonly associated with these efforts are *umbrella species* (Roberge & Angelstam 2004), *landscape species* (Sanderson et al. 2002), and *biodiversity indicators* (Caro 2010). *Focal species*, as defined by Lambeck’s (1997) original usage, have also been used to identify conservation areas (Butler et al. 2012; Nicholson et al. 2013).

Discussions about identifying and selecting important conservation areas have traditionally focused on terrestrial ecosystems; however, the process of identifying important conservation areas applies equally to marine, coastal, and freshwater ecosystems and surrogates are increasingly being used for that purpose. One example is marine protected areas (MPAs), which are marine, coastal, or estuarine areas (including intertidal areas and bays) where natural or cultural resources are given greater protection than the surrounding waters. In the US, MPAs include areas of open ocean, coastal areas, intertidal zones, estuaries, and the Great Lakes. MPAs can be designed around entire ecosystems and associated biophysical processes or they can be designed around a “focal resource,” which can be a particular habitat, species complex, single species, or cultural or other natural resource (Wenzel & D’lorio 2011).

There are many examples in the scientific literature of projects using surrogate species for defining conservation areas. **Table 3** summarizes a subset of more recent examples for each

of the various uses of surrogate species within the context of defining conservation areas. However, many of the examples have limitations and require multiple years of monitoring to assess whether expected outcomes are being achieved.

The methods for using surrogate species for identifying conservation areas have evolved over the years. Initial efforts using surrogate species to identify conservation areas focused on identifying biodiversity hotspots and broad areas that merited conservation attention (Sarkar 2014). Over time, efforts became more detailed and specific and the geographic scale decreased from global, continental and national – to regional, with a shift in focus to reserve selection and the development of complementarity approaches<sup>4</sup> (Lewandowski et al. 2010; Sarkar 2014). As this happened, the methodology also became more complex and inclusive (Moffett & Sarkar 2006; Sarkar et al. 2006). Most recently, the landscape species approach was developed. This approach combines elements from reserve selection, umbrella species selection, and general conservation planning (Sanderson et al. 2002) and is the most detailed, comprehensive, and quantitative approach for selecting conservation areas and defining key ecological processes and conservation actions within a landscape to date.

### 3.2.1 BIODIVERSITY ASSESSMENT

Biodiversity assessments are conducted for a variety of reasons. As noted in the section above, they can be used to identify areas that should be targeted for conservation actions or reserve selection at large geographic scales (global, continental, national); they can be used to monitor the health or integrity of ecosystems or landscapes; and they can be used to evaluate the effectiveness of conservation efforts or other management activities.

Regardless of the assessment's intended purpose, any attempt to include all components of biodiversity in an assessment would be effectively impossible because of data constraints, because many components of biodiversity remain largely unknown, and because of time and funding constraints. To address these constraints, biodiversity assessments rely on biodiversity indicators to fulfill their intended purpose (Howard et al. 1998; Duelli & Obrist 2003; Lamoreux et al. 2006). Biodiversity indicators are commonly used at large geographic scales to identify “biodiversity hotspots” or areas that, if protected, would prevent a large number of species from becoming extinct (Myers 1988, 1990; Myers et al. 2000), although Reid (1998) expanded the term to mean any geographic area that supports higher biotic diversity and endemism than surrounding areas and faces greater threats (Reid 1998).

For example, the 1992 Convention on Biological Diversity resulted in 20 biodiversity targets (called the Aichi Biodiversity Targets) that are designed to satisfy five strategic goals. The third of these strategic goals is to improve the status of biodiversity by

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<sup>4</sup> Complementarity approaches are defined by Lewandowski et al. (2010) as “*surrogates used to select a combination of sites that together maximize total species richness for the taxon.*”

safeguarding ecosystems, species and genetic diversity. The specific target is to have at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and integrated into the wider landscapes and seascapes by 2020 (Target 11 of the Aichi Biodiversity Targets; Secretariat of the Convention on Biological Diversity 2014). Determining whether or to what degree parties to the convention are making progress toward this target relies on biodiversity assessments commonly conducted at global, continental, or national scales. Because of the constraints on directly assessing progress toward this goal, the assessment relies on biodiversity indicators to fulfill its purpose.

These assessments have traditionally focused on centers of biodiversity or “biodiversity hotspots.” For example, Myers (1988, 1990) used lists of endemic plant species to identify diversity “hot spots.” Crumpacker et al. (1988) used potential natural vegetation to identify vegetation communities that were not well-represented on existing conservation lands; McNeely et al. (1990) used lists of vertebrates, swallowtail butterflies, and higher plants to identify countries with high levels of biodiversity; Scott et al. (1993) used spatial analyses of vegetative cover, vertebrates, and butterflies to identify areas of high biotic diversity that occurred in areas that were not represented on protected lands; while Flather et al. (1994, 1998) used county distributions of larger sets of endangered and threatened species to identify “endangerment” hotspots. Rodrigues et al. (2003) used data on the global distribution of birds, mammals, and amphibians to assess how well the global network of protected areas captured imperiled members of these taxa.

Stein et al. (2000) examined data on more than 39,000 population-level occurrences of 2,758 imperiled species, which they used as surrogates for the biodiversity of other species in the community, to identify diversity hotspots, ecological associations, and areas whose protection would be necessary to ensure the survival of these species, among other diversity metrics. Specifically, they used the system of hierarchical hexagons developed by the US Environmental Protection Agency’s Environmental Monitoring and Assessment Program to examine patterns of species occurrences and diversity hotspots. Additionally, because of their spatial scale (each hexagon is about half the size of an average county in the northeastern US) they were able to detect patterns of diversity at national and regional scales that had not been detected earlier.

Other examples include the IUCN Plant Conservation Program and Plantlife International which use endemism to identify global centers of plant diversity (“important plant areas” or IPAs); efforts to identify “key biodiversity areas” (Eken et al. 2004); and efforts by Birdlife International and the International Council for Bird Preservation to identify centers of avian endemism (Brooks et al. 2015). These are predicated on the assumption that endemic

species serve as useful surrogates of broader patterns of biodiversity. For more examples, see Radford et al. (2011).

The studies summarized thus far have used biodiversity indicators to identify areas that should be protected because of high species' richness, endemism, or because of risks to imperiled species. The use of biodiversity indicators to assess environmental conditions or management outcomes is discussed in **Section 3.3**.

**Table 3. Surrogate Species Examples for Defining Conservation: Biodiversity (Large Geographic Scale)**

Project (Source)	Summary	Comments
Amphibian biodiversity indicators in Brazil (Campos et al. 2014)	The study evaluated the performance and efficiency of eight potential indicator groups representing amphibian diversity in the Brazilian Atlantic Forest. Using MARXAN software, the performance of each indicator relative to the representativeness of amphibians in the Brazilian Atlantic Forest was verified. They found some indicators better represented the taxonomic groups assessed, than others with one group representing 71% of the amphibian species in the forest.	Study results suggest that good indicator groups can effectively represent biodiversity from a relatively small area.
Beetles and spiders as biodiversity indicators in Switzerland (Beck et al. 2013)	Investigated two levels of biodiversity indication: (1) prediction of biodiversity patterns and (2) inference of biodiversity-environment relationships. Biodiversity patterns were found to be positive but weakly correlated. Environmental models differed between both taxa and biodiversity metrics.	Need to be cautious when selecting and applying biodiversity indicators. Decisions about which metric to use are important and should be made carefully because the metric can strongly affect results.
Comparison of different diversity metrics and hotspots in arthropods in Turkey (Fattorini et al. 2011)	Analyzed spatial patterns of arthropod diversity to test whether there are multi-group hotspots or whether different groups have different areas of maximum diversification. The study used three diversity metrics: species richness, residuals from the species-area relationship, and species/area ratios. Significant positive correlations occurred for the three metrics. Advocate the use of subsets of species as surrogates for all species as long as subsets are representative of animals with different ecological needs and biogeographical histories.	Although patterns of cross-taxon diversity were significantly and positively correlated for all metrics, hotspots of different groups and using different metrics showed little overlap. Proportions of non-target species captured by hotspots of a target taxon were usually moderate.

**Table 3. Surrogate Species Examples for Defining Conservation: Biodiversity  
(Large Geographic Scale)**

Project (Source)	Summary	Comments
Mammal indicator species in Brazil (Trindade-Filho & Loyola 2011)	Examined the effectiveness and consistency of nine indicator groups in representing mammal species in two top-ranked Biodiversity Hotspots: Brazilian Cerrado and Atlantic Forest. Restricted-range species were the most effective indicators for mammal diversity as well as target species. It was also the only group with high consistency.	Showed that several indicator groups could be applied as shortcuts for representing mammal species to develop conservation plans, But only restricted-range species were consistently the most effective indicator group.
Comparison of hotspot versus complementarity approaches using biodiversity indicators (Lewandowski et al. 2010)	Reviewed published studies testing the ability of species richness of surrogate taxa to capture the richness of other taxa. Studies were stratified into two groups based on whether a complementarity approach or a richness-hotspot approach (surrogates used to select sites containing the highest species richness for the taxon) was used. They found that a surrogate was 25% more likely to be effective with a complementarity approach than with a hotspot approach.	For complementarity approaches, taxa that are feasible to measure and tend to have a large number of habitat specialists distributed collectively across broad environmental gradients (e.g., plants, birds, and mammals) were the most effective surrogates.
The projects included here are from 2010 to the present. See Chaplin et al. (2000) and Caro (2010) for summaries of earlier examples.		

### 3.2.2 LANDSCAPE DESIGN

Landscape design integrates societal values and sets biological goals by describing the landscape conditions needed to support those goals. Landscape design uses science based in landscape ecology to provide a variety of scenarios that describe where and how conservation actions can best be deployed and how those actions relate to measurable goals. Surrogate species used for this purpose can help define the conditions (e.g., habitats and their configuration, ecological processes, etc.) needed within the landscape to support populations of surrogate and beneficiary species.

For landscape and umbrella species, one or more species having the most demanding resource needs within a landscape are selected (Lambeck 1997; Caro 2010). The assumption is that by managing for those species, the needs of the less demanding species in the landscape will be met. For example, McDermid et al. (2015) used a Landscape Species Approach (LSA) to assess information on fish species in an aquatic landscape (a “freshwaterscape”) in Ontario’s Far North (OFN). The LSA methodology they applied, which was initially developed for terrestrial systems, uses five criteria — area occupied, ecological function, socioeconomic importance, habitat heterogeneity and vulnerability — to identify surrogate species in a given landscape. The OFN study analyzed 14 freshwater fish species based on the 5 criteria and selected lake sturgeon, lake trout, and walleye as landscape species because they had the highest aggregate scores. This suite of landscape species and their ecological requirements are being used to target resources for research, conservation and management within the OFN freshwaterscape.



**Figure 3.** The grizzly bear (*Ursus arctos horribilis*) is a landscape species for the Greater Yellowstone ecosystem. It was selected as one of the landscape species (Brock et al. 2006) using a quantitative approach for selecting a suite of species to conserve all habitat types, including capturing all major threats to sustainable wildlife populations within that area.



Other applications have used focal species (as defined by Lambeck 1997) to identify and manage networks of habitat types in landscapes. For example, Zack et al. (2005) used a suite of seven bird species as proxies for the more than 330 species of birds, mammals, reptiles, and amphibians dependent on oak woodlands in California for at least a portion of their life cycle. The seven species of birds were specifically selected because they represent the range of oak woodland habitats in the state.

There are many examples in the scientific literature of projects using surrogate species for defining conservation areas at the landscape scale. **Table 4** summarizes examples for each of the various uses of surrogate species within the context of defining conservation areas.

<b>Project (Source)</b>	<b>Summary</b>	<b>Comments</b>
<i>Landscape Design</i>		
Landscape design case studies: Adirondacks and Argentina (Didier et al. 2009)	Summarizes 2 of 14 landscape designs completed by the Wildlife Conservation Society. Provides detailed summary of the various steps used to complete landscape designs. Incorporates future threats into the design.	Discusses elements besides surrogate species to include in landscape design. Did not verify that selected species actually act as surrogates for beneficiary species or result in improved conservation for other species.
Comparison of two landscape design methods in Montana (Brock et al. 2006; Brock & Atkinson 2013)	Landscape design in Madison Valley, Montana. Compared two different landscape species approaches to same landscape. Both provide detailed steps.	Follow up analysis needs to be performed to ensure the scale of the design is appropriate for all species and that the design is working for all species.
Landscape design of freshwater systems in Ontario Far North (McDermid et al. 2015)	Modified the terrestrial landscape species approach to select a set of landscape species surrogates and create a landscape design for freshwater conservation for the Ontario Far North.	Data limited to only a few species of fish and results may not be applicable to smaller-bodied fish or invertebrates.
Landscape design for Michigan fishes (Esselman et al. 2013)	Landscape design using Marxan for freshwater fish conservation areas, in particular river segments, in Michigan.	Included many categories of fish and found some spatial requirement differences in the Upper and Lower Peninsula.
<i>Classic Umbrella Species</i>		
Landscape connectivity planning and surrogate species in South American temperate rainforests (Castellón & Sieving 2012)	(1) Reviewed research on landscape connectivity for the chucao and other understory-birds; (2) suggested possible strategies for scaling-up corridor designs, by addressing (meta) population viability in addition to habitat permeability; and (3) critically evaluated the degree to which these designs may or may not meet the conservation needs of other vertebrates.	Used one species to develop both a fine and large scale set of conservation areas that capture connectivity as well as habitat area requirements. Intended to only represent understory birds.

Table 4. Surrogate Species Examples for Defining Conservation Areas: Landscape Design		
Project (Source)	Summary	Comments
The example projects included here are from 2009 to the present. Previous studies are well-summarized in Caro (2010).		

### 3.2.3 RESERVE SELECTION

Reserve selection encompasses the process of identifying the location, size, and configuration of areas specifically managed to conserve species, biotic communities, ecosystems and other natural features (including geological and geomorphic features) at a local or site-specific scale. Reserves can be established in terrestrial, freshwater, coastal, and marine ecosystems.

Numerous studies have tested the use of surrogate species for identifying and selecting nature reserves. In addition to the examples presented in **Table 5**, Rickbeil et al. (2014) compared the use of environmental and habitat measures with surrogate species in regionalized schemes<sup>5</sup> for reserve selection. Their study used the North American Bird Conservation Initiative’s Bird Conservation Regions as an environmental regionalization scheme and measured the scheme’s performance by comparing it to a more detailed and data-dependent species-based regionalization. The authors concluded that the performance of the environmental regionalization schemes were almost identical to the more detailed and data-intensive approach.

Caro (2010) discussed the use of surrogate species in complementarity approaches for identifying specific reserve locations. In complementarity approaches, an algorithm is used to identify the ‘best’ reserve site and then the next best site following an iterative, step-wise process. The goal of these approaches is to capture the ‘most’ species across a reserve network. This particular use for surrogate species is one of the better studied and there are many examples in the literature (Nicholson et al. 2013; Neeson et al. 2013; Olds et al. 2014; Di Minin & Moilanen 2014; Myšák & Horsák 2014; Rickbeil et al. 2014); a few are summarized in **Table 5**.

Table 5. Surrogate Species Examples for Defining Conservation Areas: Reserve Selection		
Project (Source)	Summary	Comments
Red-cockaded woodpecker and habitat	Red-cockaded woodpecker and one butterfly and one frog were assessed to	Based on modeling, red-cockaded woodpecker was not a

<sup>5</sup> Regionalizations divide geographic regions into areas of relatively similar environmental characteristics. Per Rickbeil et al. (2014) regionalization schemes are “assumed to delineate distinct ecological communities of target species” and “are commonly constructed using surrogates.”

corridors (Breckheimer et al. 2014)	determine whether any functioned as suitable surrogate species for determining dispersal habitat. The frog was the most useful umbrella species.	good surrogate. This was not based on field results after a conservation program had been implemented.
Surrogates for diversity and reserve selection for Mediterranean herpetofauna (Carvalho et al. 2011)	Used species distribution patterns and environmental gradients as surrogates for genetic diversity. Used biotic elements and environmental categories as surrogates for the neutral and adaptive components of genetic diversity, respectively. The study identified nine biotic elements. Priority areas selected in the three scenarios were similar in area amount but had low spatial agreement.	Prioritization exercises that integrate surrogates for evolutionary processes can deliver spatial priorities that are fairly different to those where only species representation is considered.
Charismatic megafauna and other surrogates for reserve selection in Africa (Di Minin & Moilanen 2014)	A total of 662 biodiversity features, including habitat types, and species and populations were used. Other taxa are not good surrogates for charismatic mammals. Habitat types are a necessary component of surrogacy strategies that cover plants and insects. A combination of habitat types and charismatic mammals, complemented with other well-known taxa, provided the highest surrogacy effects. Generally, the low capacity of one taxon to predict priority areas for other taxa was confirmed.	Also includes elements of use as flagship species and use of surrogate habitat information.
Evaluating a surrogate species and the resulting marine reserve design (Olds et al. 2014)	Looked at marine reserve design and effectiveness. They tested the effectiveness of implemented reserve designs based on the bumphead parrotfish as a surrogate species (umbrella species). Actual field verification that creation of reserves designed based a surrogate species benefitted other species. If only isolated reefs were examined, the conclusion may have been different.	Field test of reserve design using a surrogate. One of few studies to confirm results of reserve design relative to beneficiary species.
Evaluation of higher taxon surrogates (Neeson et al. 2013)	Developed a mathematical model to show how taxonomic diversity, community structure, and sampling effort together affect three measures of higher taxon performance: the correlation between species and higher taxon richness, the relative shapes and asymptotes of species and higher taxon accumulation curves, and the efficiency of higher taxa in a complementarity-based reserve-selection algorithm.	The found that higher taxon surrogates performed well in communities in which a few common species were most abundant, and less well in communities with many equally abundant species.
The examples projects included here are from 2010 to the present. Previous studies are well-summarized in Caro (2010).		

Mammalian carnivores have tended to be selected as “area-demanding umbrella species” because they have large home ranges and require large tracts of habitat (Eisenberg 1980; East 1981; Peterson 1988; Shafer 1990; Noss et al. 1996; Carroll et al. 2001). However, herbivores (Mealy & Horn 1981; De Vries 1995; Berger 1997) and birds (Martikainen et al. 1998; Suter et al. 2002) have also been used for this purpose. Bird and insects have tended to be selected as “site-selection umbrella species” and as “extended umbrella species” (Roberge & Angelstam 2004). Often different groups of surrogate species are compared to see if the reserve networks identified are congruent across the various surrogate groups (Caro 2010). The results from a number of studies examining this issue are mixed (Caro 2010; Nicholson et al. 2013).

#### 3.2.4 ROLE OF ENVIRONMENTAL SURROGATES

Environmental surrogates (defined in **Section 2.8**) are sometimes used instead of or in conjunction with surrogate species. Di Minin & Moilanen (2014) examined multiple possible surrogate taxa and habitat type data, in combination with charismatic megafauna, for reserve selection using complementarity in southern Africa. They found that including habitat types greatly increased the representation of otherwise poorly represented taxa.

Surrogate habitat measurements are often too coarse for developing specific local or regional landscape conservation designs but are useful for other landscape-scale conservation efforts, such as identifying areas of rapid change or using surrogate habitat as a proxy for priority species population data. Some studies indicate that habitat measurements may be used in conjunction with surrogate species during comprehensive planning efforts (Carmel & Stroller-Cavari 2006; Carvalho et al. 2011; Schindler et al. 2013; Lindenmayer et al. 2014; Di Minin & Moilanen 2014).

### 3.3 USING SURROGATE SPECIES AS INDICATORS OF MANAGEMENT AND/OR ENVIRONMENTAL CONDITIONS

The use of surrogates as indicators of management and environmental conditions has a long history. Surrogates have been used to assess land for potential grazing and agricultural use (Shantz 1911; Clements 1916), and to monitor air pollution (Bobrov 1955; Middleton 1956; Weinstein & McCune 1970), water pollution (Cairns, Jr et al. 1973; Cairns, Jr 1977), and other toxins for more than half a century. Moore (1962) used ten “indicator species” to describe changes in heathland habitat in south western England, assess the effects of fragmentation on the habitat, assess its current status, forecast future patterns, and develop conservation recommendations. Moore (1996) discussed the importance of using indicator species to monitor contamination.

Since these early applications, surrogates have been extensively enlisted as indicators of management action and environmental conditions. The applications are too extensive for this document, but see **Table 6** for a sample of applications and see Hammond et al. (1995), Hák et al. (2007), Jørgensen et al. (2010a), and the journal *Environmental Indicators* for more complete coverage of the various applications that have employed surrogate species and other surrogate measures. The remainder of this section provides a brief summary.

Despite this long history, the use and reliability of surrogates as indicators of management and environmental conditions continues to be debated (Carignan & Villard 2002; Andersen & Majer 2004; Morrellet et al. 2007; Regan et al. 2008; Fleishman & Murphy 2009). However, a growing body of literature supports the use of surrogates for this purpose (e.g. Rice 2000, 2003; Kurtz et al. 2001; Carignan & Villard 2002; Manne & Williams 2003; Rice & Rochet 2005; Rochet & Rice 2005; Bani et al. 2006) and several frameworks for selecting appropriate indicators have been proposed (Pannell & Glenn 2000; Dale & Beyeler 2001; Niemeijer & de Groot 2008; Smyth et al. 2009).

For many of these applications, surrogate species are used in conjunction with conceptual models that have been constructed to assess the risks management or environmental conditions pose. For these applications, surrogates are “health indicators” or “population indicators” (Caro & O’Doherty 1999) or “trend indicators” (Gregory et al. 2005) as opposed to indicators of biodiversity. Surrogates that are used as indicators of trends are used to inform the “state” variables in pressure-state-response and driver-pressure state-impact-response assessment models (Gabrielsen & Bosch 2003; Sherman et al. 2005; Labiosa et al. 2014).

For other applications, surrogate species are used as part of multi-metric indices that are used to: (1) classify environments; (2) select measurable attributes that provide reliable and relevant signals about the biological effects of management action or of other human activities; (3) to support monitoring; and (4) to communicate this information to the public and policymakers (Karr 1981, 2006; Karr & Chu 1997). These indices, including the widely-used Index of Biotic Integrity or “IBI,” are generally dominated by metrics of taxonomic richness (number of taxa) because shifts among taxa are more sensitive to levels of environmental stress than to changes in ecosystem processes. Multi-metric indexes, like the IBI, integrate multiple biological indicators (e.g. taxa richness, indicator taxa or guilds, health of individual organisms, and trophic structure or reproductive biology) to measure the condition of a complex system (Karr 2006). These indices are designed to discriminate between the “signal” of an environmental stressor and the “noise” of natural variability (Karr & Chu 1997; Karr 2006).

The U.S. Forest Service and Bureau of Land Management have used management indicator species in several broad-scale, ecosystem-based land management strategies. Examples

include the Northwest Forest Plan (USFS & BLM 1994; Rapp 2008; Davis et al. 2011), the Southern Appalachian Assessment (SAMAB 1996), the Sierra Nevada Assessment (Erman & Science Team 1997; USFS 2014), and the Interior Columbia River Basin Ecosystem Management Project (Wisdom et al. 2000). In the latter of these efforts, the USFS focused on “broad-scale species of focus,” which were vertebrate species whose population size was known or suspected to be declining in response to habitat conditions or other human activities and whose habitats could be reliably estimated using large-scale mapping and spatial analysis (Wisdom et al. 2000). However, in 2012, the U.S. Forest Service promulgated new planning regulations that diminished the traditional role of management indicator species and focused its management activities on habitat as an indicator of ecosystem health (summarized and critiqued in (Schultz et al. 2013).

Surrogates are also used as global and continental indicators of ecological health. Examples of this application (biodiversity assessment) include the 10 Living Planet Reports prepared by the World Wildlife Fund since 1998. Living Planet Reports measure trends in global biological diversity by tracking populations of 1,313 vertebrate species, sub-divided into terrestrial, marine, and freshwater species groups (World Wildlife Fund 2014). Trends associated with these species are assumed to be representative of overall trends in global biodiversity and ecosystem health. Other examples include the Global Biodiversity Outlooks, which measure progress toward the Aichi Biodiversity Targets associated with the 1992 Convention on Biological Diversity (Secretariat of the Convention on Biological Diversity 2014).

Surrogates are used as indicators of ecological health in marine and coastal ecosystems as well. For example, management of large marine ecosystems (LMEs) — regions encompassing 200,000 km<sup>2</sup>, or larger, that are characterized by distinct bathymetry, hydrography, productivity, and tropically-dependent populations — generally rely on a suite of indicators of productivity, fish and fisheries, pollution and ecosystem health, socioeconomics, and governance to measure the LMEs’ changing states (Sherman et al. 2005; Spalding et al. 2007; Sherman & Hempel 2009). These management activities combine biodiversity indicators with a driver-pressure-state-impact-response system to support adaptive management actions (Sherman et al. 2005). For example, the indicators used in the fish and fisheries module — biodiversity, finfish, shellfish, demersal species, and pelagic species — are used to support ecosystem-based management decisions associated with commercial fisheries (Sherman & Hempel 2009).

**Table 6. Surrogate Species Examples for Indicators**

Project (Source)	Summary	Comments
<i>Management Indicator</i>		
Selecting indicators	Developed an approach that applies	Monitoring programs that

**Table 6. Surrogate Species Examples for Indicators**

Project (Source)	Summary	Comments
species for evaluating predator management (Tulloch et al. 2013)	decision science and selects the best complementary suite of species to monitor to meet specific conservation objectives. Created an index for indicator selection that accounts for the likelihood of successfully detecting a real trend due to a management action and whether that signal provides information about other species. Their method selected the species that provided more monitoring power at lower cost relative to traditional approaches that consider only a subset of the important considerations.	ignore uncertainty, likelihood of detecting change and complementarity between species will be more costly and less efficient and may waste funding that could otherwise be used for management.
Assessing recovery after deer density reduction (Bachand et al. 2014)	Used plants, carabid beetles, bees, moths and songbirds to assess which ones best indicated responses to deer density management. Multi-species sets were better at representing responses and plant/moth combinations provided the best coverage.	Excellent study design for identifying appropriate management indicator species for a larger region/landscape.
Forest birds as indicators for exotic mammal management (Hoare et al. 2013)	No support for using forest birds (21 species) as management indicator species for responses to exotic mammal management.	Limited number of species and limited correlations among species with respect to the respond to management.
Selection of indicator(s) for monitoring the success of management actions (Tulloch 2011)	The new quantitative cost-effectiveness (benefit/cost) approach developed here will allow transparent, explicit, credible, accountable selection of indicator species that can demonstrate improved performance of environmental management programs and show that money has been effectively used to produce environmental benefits.	
<i>Environmental Indicator</i>		
Indicator species for monitoring mercury in Great Lakes (Evers et al. 2011)	Walleye, largemouth bass, lake trout and herring gull eggs in the Great Lakes Region were monitored for concentrations of mercury as indicators of mercury pollution in the environment and water, which are difficult to measure directly. The assumption being that the mercury affecting these species will have similar effects on other species in this ecosystem.	No validation of surrogacy assumption
Lichens and mosses as indicators for climate change in Alaska (Wesser 2011)	Lichens and mosses are monitored as indicators of climate change, air quality and wildlife (caribou) habitat in Alaska and the Pacific Northwest.	Data collected over many years and used for a variety of purposes.
National Ecological Observatory Network	NEON is planning long-term, continental-scale monitoring of species to ascertain the	Data is available for use but not collected specifically to

**Table 6. Surrogate Species Examples for Indicators**

Project (Source)	Summary	Comments
(NEON) program (Kao et al. 2012)	effects of climate change.	address a particular conservation objective.
Additional examples prior to 2010 are provided in Caro (2010).		

### 3.4 USING SURROGATE SPECIES TO ENGENDER PUBLIC SUPPORT

Iconic or charismatic species are commonly used as flagship species to gain support from the public or stakeholders for landscape conservation efforts (Caro & O’Doherty 1999; Caro 2010; Barua et al. 2011; Veríssimo et al. 2011). Historically, flagship species were rarely intended or verified to be surrogates but were used primarily as tools to raise funds and sometimes public support (Caro 2010). Flagship species are only in a surrogate role if by raising awareness of the flagship species, public appreciation grows for the landscape and/or the species it supports and improved conservation outcomes for both the flagship and the beneficiary species are realized (Caro 2010; Barua et al. 2011; Veríssimo et al. 2011).

Earlier studies using flagship species selected species based on their marketing value and ecological significance. More recent studies recommend selecting flagship species based only on their ability to gain public support and using marketing techniques to tailor species selected toward defined target audiences (Abeyta et al. 2011; Veríssimo et al. 2011, 2014a, 2014b; Nekaris et al. 2015).

One recent study tested the application of “Cinderella species” which are “esthetically pleasing and overlooked species that fulfill the criteria of flagships or umbrellas” (Nekaris et al. 2015). Multiple criteria were used to select a less charismatic species that was pleasing to the public (Cinderella species) but also fulfilled umbrella species criteria. A survey questionnaire was distributed to the public using a short list of species and species distributions were modeled. The method resulted in the selection of two species, the red slender loris and fishing cat; both species were appealing to the public and fulfilled ecological criteria.

In another study a systematic and stakeholder-driven approach was used to select a new flagship species using environmental economic and marketing techniques in Brazil (Veríssimo et al. 2014b). The results of the new methodology ranked the previous flagship species at 142 out of 221 possible bird species, confirming that it was indeed not a good flagship. The authors concluded that using a systematic and stakeholder-driven approach to select flagship species allowed for identification of flagship species most likely to be successful at gaining public support.



Table 7 presents a subset of studies from the recent literature focused on flagship species.

<b>Table 7. Surrogate Species Examples for Flagship Species</b>		
<b>Project (Source)</b>	<b>Summary</b>	<b>Comments</b>
Flagship Bird Species Selection for SAVE Brasil (Veríssimo et al. 2014b)	The project used a systematic and stakeholder-driven approach to select flagship species for a conservation campaign in the Serra do Urubu in northeastern Brazil, because a previous flagship was not functioning as expected. This approach was compared with selection via a plurality flagship vote.	Did confirm that existing flagship species is not a good choice.
Araripe manakin flagship program in Brazil (Veríssimo et al. 2014a)	Evaluated current flagship and the various target audiences in Brazil to determine if an additional flagship species was necessary. Results indicated existing flagship campaign is raising public awareness but one additional flagship may be merited based on target audience characteristics.	Good example of evaluating effectiveness of flagship and identifying modification to improve success.
Selecting a surrogate species serving dual role of flagship and umbrella species in Sri Lanka (Nekaris et al. 2015)	Selected taxa that fulfilled both flagship and ecological roles. Created a shortlist of species and from a survey of local perceptions highlighted two finalists. Tested for umbrella characteristics against the original shortlist and selected two species – one had higher flagship potential.	Study goal was to select for multiple criteria and select a species that was a trade-off between conservation goals and appeal to the public.
Charismatic megafauna and other surrogates for reserve selection in Africa (Di Minin & Moilanen 2014)	While the primary focus of the research was on reserve design, the inclusion of charismatic megafauna highlighted their value as flagships as well as umbrella species. Notable results include that other species did not represent charismatic megafauna well and, conversely, that the megafauna poorly represented other species, other than mammals. See Table 5.	They did not examine how well the identified flagship species served as surrogates for raising public awareness and public support.

## 4 SELECTING SURROGATE SPECIES

### 4.1 INTRODUCTION

Selecting a useful surrogate from a suite of candidates is one of the greatest challenges to conservation efforts that rely on them and requires the use of clearly defined criteria. The criteria for selecting surrogate species can vary widely among applications and are the subject of extensive research and development. This section summarizes some of the selection criteria that have been published and focuses on criteria that are common to most published selection processes. For detailed discussions of the different approaches for developing criteria, readers should refer to Sanderson et al. 2002; Lindenmayer & Likens 2011; Suring et al. 2011; Veríssimo et al. 2011; Epps et al. 2011; Brock & Atkinson 2013; and Ruaro & Gubiana 2013. Furthermore, given the evolving nature of the science of surrogates, these criteria should be viewed as works in progress.

Many criteria lists include a criterion similar to ‘feasible to monitor’. This stems from the assumptions that it should be possible to document changes in a surrogate species population and that a surrogate species should be cost-effective to monitor (Caro 2010). If it is hard to document changes in a population (either because the species is hard to monitor or it is hard to get enough funding to monitor it) then the species may not provide enough data to evaluate the success of the conservation program and make decisions with any regularity. However, there are circumstances where feasibility to monitor the surrogate will not be important. For example, surrogate species may be selected to help design a landscape but other species may be monitored to test whether or not the conditions in the landscape support the larger pool of species of conservation interest. In general, the criteria used for selecting surrogates should be rigorous and systematic – and specific to the intended use of the surrogate.

### 4.2 CRITERIA FOR SELECTING SURROGATE SPECIES FOR DEFINING CONSERVATION AREAS

The criteria for selecting surrogates that can be used to identify, select, and rank conservation areas are strongly influenced by the spatial scale of the analysis. At global, continental, and national geographic scales, the surrogates typically selected are biodiversity indicators (see **Section 4.3**). When the geographic scale is regional or local, for example when identifying and selecting reserves or designing landscapes, biodiversity indicators become increasingly unreliable as surrogates for other species (for example, see Ekroos et al. (2013) and de Andrade et al. (2014)). At regional and local spatial scales, landscape species and umbrella species perform better as surrogates (for example, see Breckheimer et al. (2014); de Andrade et al. (2014)). This section focuses on umbrella species landscape species and Lambeck’s focal species,

## *Criteria for Species for Defining Conservation Areas (Regional/Local)*

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### *Common to Focal, Umbrella, and Landscape Species*

- Species are representative
- Use large areas
- Occupy a variety of habitats
- Vulnerable to environmental stressors
- Feasible to monitor\*

### *Focal Species (Lambeck 1997)*

- Most demanding of resources/area (limited by area, dispersal, resources or processes)

### *Umbrella Species*

- Well-known natural history and ecology
- High probability of population persistence

### *Landscape Species*

- Requires processes and ecological functions at the landscape level
- Social and economic significance

\*The importance of this criterion will be dependent upon the goal the surrogate is being used to help achieve (see Section 4.1).

Compiled from Coppolillo et al 2004; Seddon & Leech 2008; Brock & Atkinson 2013

In addition to ensuring that surrogates used to define conservation areas are representative of beneficiary species, communities, habitats, and ecosystems, several other considerations must be made. One consideration is the number of surrogates that will be selected: multiple species or sets of species generally perform better than single species (Darwall & Vié 2005; Butler et al. 2012; Nicholson et al. 2013; Breckheimer et al. 2014; de Andrade et al. 2014).

Another consideration is which taxa to select. A wide variety of taxa have been used as surrogates, but the most common practice is to select well-defined taxonomic groups because they require less taxonomic expertise than less-studied taxa. In addition to mammals and birds, investigators have reported that butterflies (Mac Nally & Fleishman 2004; Lovell et al. 2007), beetles (Báldi 2003; Lovell et al. 2007; Azeria et al. 2009), vertebrates (Lund & Rahbek 2002) and vascular plants (Kati et al. 2004; Anand et al. 2005; Nordén et al. 2007) have been effective surrogates in many different terrestrial settings.

A related is the number of taxa that will be included in the set of surrogates. Heino (2010) argued that single groups are ineffective as predictors of variation in the biotic diversity of other taxonomic groups. In response, several investigators have selected species from a small group of different taxa as surrogates (Darwall & Vié 2005; Butler et al. 2012;

Nicholson et al. 2013; Breckheimer et al. 2014; de Andrade et al. 2014) and reported that the results were reasonably representative of beneficiary species, communities, and ecosystems.

The final consideration is the characteristics or attributes of the surrogate species. To select landscape species, several authors recommended selecting species that (1) use large areas (i.e., have large home ranges); (2) occupy a variety of habitats; (3) are vulnerable to environmental stressors in the landscape; (4) require processes and ecological functions that operate at the landscape level; and (5) are considered culturally or economically significant (Sanderson et al. 2002; Coppolillo et al. 2004; Brock & Atkinson 2013).

To select umbrella species, Seddon & Leech (2008) recommended using large home range as a criterion. They also recommended a number of other characteristics be considered including that surrogate species have well-known natural histories and ecologies; high probabilities of population persistence; co-occurrence with other species; that their management benefits other species; that they have moderate sensitivity to human disturbance; and that they are easily sampled or observed.

When sets of species are used as surrogates, investigators have also recommended using community-level metrics like indices of similarity and dissimilarity (Spellerberg 2005; Heino 2010) and formal statistical models to ensure that the set of species selected as surrogates are representative of larger communities.

### 4.3 CRITERIA FOR SELECTING SURROGATES AS BIODIVERSITY INDICATORS

Because biodiversity encompasses so many different phenomena and levels of biotic organization, the criteria used to select biodiversity indicators will depend on the aspect of

#### *Criteria for Biodiversity Indicators*

- Represents other species or some other ecologically important feature
- Species at risk
- Restricted range
- Biome restricted species sets
- Species that congregate during a life stage
- Feasible to monitor
- Available data
- High cross-taxon congruence

Compiled from Duelli & Obrist 2003; Eken et al. 2004; Caro 2010

biotic diversity the indicator is intended to represent and how the indicator will be used (Duelli & Obrist 2003). Surrogates selected as indicators of genetic diversity will not be the same as surrogates selected as indicators of species or ecosystem diversity. Similarly, surrogates selected to identify unprotected areas with high species diversity may not be the same as surrogates selected to monitor the health, sustainability, or resilience of an ecosystem, watershed, region, continent, or the planet. Regardless of the aspect of biodiversity being measured and the purpose of taking the measures, the surrogates are intended to convey meaningful information about something larger than the surrogate itself (Duelli & Obrist 2003; Biodiversity Indicators Partnership 2011).

Many of the early biodiversity assessments relied on a “critical faunas analysis” to select indicators of biodiversity. The term was introduced by Ackery & Vane-Wright (1984) and originally focused on a single taxonomic group of narrow endemics (called a “complement”) to identify the minimum number of areas that would be necessary to support at least one population of every species in the “complement.” The single most important site for conservation would be the one that supported the largest proportion of the “complement.”

Myers and colleagues added degree of threat as a criterion for identifying biodiversity indicators (Myers 1988, 1990; Myers et al. 2000). Eken et al. (2004) expanded the “endemism” criterion to include species with restricted ranges and included (1) globally threatened species, (2) congregations of species that concentrate at particular sites during some stage in their life cycle, and (3) biome-restricted species assemblages. These criteria are commonly used to select biodiversity indicators.

At global, continental, and national spatial scales, biodiversity indicators tend to be selected using two criteria: (1) the ease of monitoring their populations and (2) the availability of data on distribution and abundance. Most uses of biodiversity indicators tend to rely on any taxa that meet these criteria. For example, Myers (1988, 1990 and Myers et al. 2000) selected vascular plants, mammals, birds, reptiles, and amphibians (excluding fish and invertebrates) to identify “biodiversity hotspots.” Crumpacker et al. (1988) selected vegetative communities as biodiversity indicators while Scott et al. (1993) used a combination of vegetative cover, vertebrates, and butterflies. Stein et al. (2000) went even further by using data from State Heritage Programs to examine the distribution, ecological associations, and threats facing 2,758 imperiled species as well as species that were listed as endangered or threatened.

#### 4.4 CRITERIA FOR SELECTING SURROGATES AS MANAGEMENT OR ENVIRONMENTAL INDICATORS

Criteria for selecting indicator species have a long history. Moore (1962), who is reported to have been the first to explicitly use “indicator species,” used 10 “indicator species” to describe changes in heathland habitat in south western England, assess the effects of fragmentation on the habitat, assess its current status, forecast future patterns, and

develop conservation recommendations. Moore (1962) used a species' habitat specificity as his selection criterion: he paired a species that had an obligate relationship to the habitat with a species that had a facultative relationship and used their relative abundance to

### *Criteria for Management and/or Environmental Indicator Species*

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- Feasible to monitor
- Sensitive to the change of interest
- Response to change is known, predictable, and have low variability
- Response to change should be specific and not influenced by other factors
- Response of indicator should represent beneficiary species
- Response can be anticipatory (only relevant for some conservation objectives)

“indicate” the quality of heathland habitat.

Like Moore (1962), Hill et al. (1975) used the term “indicator species” in a methodology designed to classify vegetative communities. They defined a “perfect” indicator as a species that has a completely specific (obligate) relationship to a particular community: it occurs in all communities of a particular type and never occurs in other communities. They argued that “a species has no indicator value at all if it” does not provide this function; nevertheless, they presented a formula for calculating a species' indicator value that scaled from zero (no indicator value) to one (perfect indicator value). Applying this principle more generally would mean that a species' indicator value can be measured by its degree of sensitivity to a specific management regime or environmental condition.

Odum (1971) echoed these criteria by arguing that species with narrow environmental tolerances (he used the term “steno species”) make much better indicators than species with wide environmental tolerances (“eury species”), particularly if the former have strong fidelity to a site, habitat type, or ecosystem. He also argued that large species make better indicators than small species because they are more stable and have lower turnover rates than smaller species. Finally, he argued that numerical relationships between species, populations, and entire biotic communities are often more reliable indicators of environmental conditions than single species because “wholes” (populations and biotic communities) will be better at integrating environmental conditions than “parts” (single species).

More recently, procedures for selecting both management and environmental indicators have focused on selecting species that are responsive to change – either change in management or environmental condition – along with the rest of the criteria normally considered for management and environmental indicators. Ideally, rather than a single indicator, a suite of indicators provides coverage across the range of potential change being assessed (Dale & Beyeler 2001; Niemeijer & de Groot 2008; Tulloch et al. 2011). The criteria listed in the text box can be broken down into more specific components to meet a specific monitoring purpose (Niemeijer & de Groot 2008; Tulloch et al. 2011).

With two exceptions, these criteria closely correspond to the criteria for management indicator species established in the National Forest Management Act. The two exceptions are species listed as threatened, endangered, or rare and species having social or economic value. Although US Forest Service planning processes have not been required to designate management indicator species since 2005, the concept is still used by organizations because it allows land managers to make decisions in the face of limited resources (for example, see Moseley et al. 2010).

A common approach to selecting indicators is to identify a suite of potential indicators and rank them using a set of pre-determined criteria (Rice & Rochet 2005; Tulloch et al. 2011, 2013). There are many suggested methods for this aggregation, each with its own informational requirements and mathematical properties (Figueira et al. 2005; Moffett & Sarkar 2006). However, due to problems inherent in aggregation and potential for subjectivity in scoring (Wolman 2006; Steele et al. 2009), some authors have suggested that indicator species are most useful if their selection is primarily based on a sound quantitative database from the focal region rather than qualitative criteria such as those commonly used for scoring approaches. Empirical analyses can then be used to investigate the trade-off between criteria such as species monitoring cost, ease and feasibility, responsiveness, and representativeness, to select an indicator species that is both cost-effective and informative. In some cases, the surrogate is selected as an indicator because its appearance, density, or dominance in a community or ecosystem serves as evidence of ecosystem degradation (Jørgensen et al. 2010b).

Butler et al. (2012) started with Lambeck's original idea of "focal species" and developed criteria for selecting indicator species. First, they articulated two principles the set of indicators would have to satisfy: (1) all resource types used by the wider community (the beneficiary species) must be exploited by at least one of the focal species and (2) the set of focal species must be comprised of the most specialist species possible. The first of these two principles ensures that the ecological niche species occupied by the focal species encompasses the niche space of the beneficiary species. The second of these principles is designed to maximize the sensitivity of the focal species (members of the set of species) to ensure that they serve as an early warning system for the beneficiary species.

Butler et al. (2012) identified appropriate indicators by first identifying a pool of potential indicator species. They then constructed a matrix of resource requirements for the species in the pool and assessed the resource dependencies to ensure they encompassed the full range of resources available in the area being assessed. Then they examined different combinations of species in the set (identifying all two-species combinations, three-species combinations, etc.) to identify those combinations that contained species that exploited the full range of resources in the area being assessed. Finally, they evaluated the sensitivity of the sets of species to identify the optimal combination that satisfied their two principles.

#### 4.5 CRITERIA FOR SELECTING SURROGATE SPECIES AS FLAGSHIP SPECIES

The purpose of a flagship species is to either increase public support for a specific conservation objective or to raise funds for one or more conservation objectives, which benefit other species. Various combinations of ecological, phenotypic, cultural and policy related traits have been recommended for selecting flagship species (Dietz et al. 1994; Caro & O'Doherty 1999; Bowen-Jones & Entwistle 2002; Barua et al. 2011; Veríssimo et al. 2011). The list of criteria can vary depending upon the conservation objective and target audience. A number of recent publications have tested and proposed new methodologies, incorporating techniques from market research (Barua et al. 2011; Veríssimo et al. 2011, 2014a, 2014b; Kanagavel et al. 2014; Nekaris et al. 2015).

##### *Criteria for Flagship Species*

- Occurs in the geography of interest
- Appropriate conservation status and population size
- Represent other species, preferably with an umbrella effect
- Recognizable and easily observed
- Cultural significance and positive associations for the target audience
- Traditional knowledge and common names
- Economic value
- Scientific value
- Charismatic
- Existing use

Compiled from Barua et al. 2011 (not all criteria apply in all circumstances)





**Figure 4. The Seven-colored Tanager (*Tangara fastuosa*) is a flagship species for the forest birds of Brazil.** Like the Monarch butterfly in North America, its striking coloration make it charismatic; however, its selection was systematic, transparent and stake-holder driven (Veríssimo et al. 2014), and the tanager species was selected for a specific conservation campaign in the Serra do Urubu in northeastern Brazil.

## 5 THE VALIDITY OF THE SURROGACY ASSUMPTION

### 5.1 INTRODUCTION

Any conservation effort that relies on a surrogate makes two fundamental assumptions: (1) the surrogate is an appropriate representative of a larger group of species, biotic communities, or ecosystems and (2) conservation objectives can be achieved via the use of surrogates. This section focuses on the first of these assumptions while **Section 6** focuses on the second assumption.

Although surrogates have been used for decades, recent studies have devoted more attention to testing the assumption of representativeness than earlier studies. The studies that have evaluated the assumption of the representativeness of surrogates have essentially tested the validity of the criteria used to select the surrogate (discussed in **Section 4**). Many of the studies that evaluate the representativeness of a surrogate use the term “congruence,” which Gaston (1996) originally defined as the “degree to which those areas in which the diversity (usually the number of species) of a given taxon attains high values...coincide with those areas in which one or more other taxa attain high diversity values.” Today, “congruence” is used more generally to mean the degree to which areas given a high value based on surrogate species would also be given a high value for the beneficiary species (for example, see Butler et al. 2012).

The studies discussed in this section have used different approaches to test the assumption that a surrogate is an appropriate representative of a larger group of species, biotic communities, and ecosystems. However, most of these studies have focused on a common set of variables to test this assumption: (1) the number of surrogates that are selected; (2) the taxa that are selected as surrogates; (3) the method used to select the surrogate; (4) the attributes of surrogates the method relied on; and (5) the species, communities, and ecosystems the surrogates are used to represent. This section discusses the various studies that have tested the representativeness assumption of the different kinds of surrogates, the variables tested, conclusions about the assumption, and any limitations investigators identified that were necessary to ensure the assumption of surrogacy was valid.

A number of earlier investigators raised concerns about the conceptual, theoretical, and practical basis of taxon-based surrogate schemes such as umbrella species (e.g., Landres 1983; Simberloff 1998; Andelman & Fagan 2000; Lindenmayer et al. 2000). These published critiques of taxon-based surrogate schemes apply to focal, indicator, and umbrella species. For example, (Lindenmayer et al. 2002) invoked a suite of theoretical reasons to argue that taxon-based surrogate schemes generally could not be considered representative (citing Landres 1983; Morrison et al. 1992; Simberloff 1998; Gascon et al. 1999; Andelman & Fagan 2000; Lindenmayer et al. 2000).

Favreau et al. (2006) and Lindenmayer & Likens (2011) expressed similar reservations about the theoretical foundations of the umbrella species and focal species concepts. These authors argued that taxon-based surrogate schemes assume that the response of particular species will be indicative of the response of many other species. They argued that this assumption was not valid, because the effects of landscape change and habitat fragmentation could vary among species (e.g., Robinson et al. 1992) and among groups of species (e.g., Gascon et al. 1999). Members of the same guild may not respond in the same way to a given type of disturbance, even when they are closely related (Landres et al. 1988; Morrison et al. 1992).

In his response to Lindenmayer's 2002 critique of the focal species approach, Lambeck (2002) states that the critique "was built on unsound logic" and "misleading assumptions." He further clarifies that a focal species approach does not assume that the response of particular species will be indicative of the response of many other species and is not expected to result in patterns of nested occurrences or nested responses. He reiterates that the focal species assumption that multiple threats act differentially on different species provides a good argument for not expecting to see nested responses or nested occurrences of species with a focal species approach.

Most of the studies referenced in the preceding paragraph were based on studies using single surrogate species as opposed to groups of surrogate species. Caro's (2010) summary of more recent studies reported better success when groups of surrogate species were used, leading him to conclude that, while the results are not conclusive, they are promising.

## 5.2 TO DEFINE CONSERVATION AREAS

As noted by Nicholson et al. (2013), most testing of surrogate species for landscape design is based around species distribution and not species persistence (and therefore overlaps with research on biodiversity indicators). Testing the assumption that a surrogate species (or set of species) do indeed represent other species' distribution is an important first step for using and/or selecting a surrogate species (or set). A priori testing of the assumption that the persistence and/or abundance of a surrogate species represents the persistence and/or abundance of other species is more difficult, but more relevant to the conservation task of designing a landscape.

Several studies have argued that measuring the performance of surrogates requires selecting notional conservation areas based on a surrogate (or surrogates) and then measuring species representation and then comparing these results with the results obtained from conservation areas generated at random (Brooks et al. 2004; Rodrigues & Brooks 2007; Grantham et al. 2010). This is a method of validation that has rarely been used.

For example, Rodrigues & Brooks (2007) state: “The relevant question in a surrogacy test is, therefore, what is the extent to which areas selected for surrogates capture the target features?” Rodrigues & Brooks (2007) present arguments about why there is not a clear single, effective method for answering this question. They argue that the respective assumptions, strengths and limitations of selection-based and pattern-based testing methods remain poorly understood; different plausible methods produce different and sometimes conflicting results; and the data-dependence of results from different methods has been poorly explored by applications to multiple regions and planning situations.

Lawler & White (2008) report that, in general, studies testing the assumption of surrogacy related to defining conservation areas have “produced diverse and often contradictory results.” They write that it is difficult to compare across studies because the studies differ in spatial scale, in the surrogate groups used, and in the methods; while some authors report a positive relationship between surrogate groups and biodiversity (specific to the goal of maximizing biodiversity within a conservation area), generalizations about what makes a surrogate perform well cannot be made.

The most recent trend in validating assumptions for reserve selection or landscape design is to generate several different potential outcomes using different scenarios and then use models to compare how well each outcome ‘represents’ the different beneficiary species (Carvalho et al. 2011; Epps et al. 2011; Cushman & Landguth 2012; Larsen et al. 2012; Nicholson et al. 2013; Lindenmayer et al. 2014; Di Minin & Moilanen 2014). For example, Di Minin & Moilanen (2014) evaluated both habitat and species biodiversity surrogates for reserve selection. They developed two spatial conservation priorities – one starting with charismatic megafauna and one starting with habitats, then supplemented each with additional information. Their results indicate that other taxa are not good surrogates for charismatic mammal species and that habitat types are a necessary component of surrogacy strategies that cover plants and insects. Overall, a combination of habitat types and charismatic mammals, complemented with other well-known taxa (birds, amphibians and reptiles), provided the most complete coverage and best results for identifying conservation areas. As with other studies, these results indicate the need to use more than one surrogate species.

### 5.2.1 UMBRELLA SPECIES

Roberge & Angelstam (2004) described and evaluated three variants of the umbrella species concept: area-demanding umbrella species, site-selection umbrella species, and an “extended umbrella concept.” The first two of these variants correspond to the “classic umbrella species” and “local umbrella species” recognized by Caro (2010) and Zacharias &

Roff (2001), while the third of these variants overlaps with the focal species concept, as envisioned by Lambeck (1997).

Roberge & Angelstam (2004) reviewed 110 published articles, chapters in books, papers from conference proceedings, and “grey literature” that discussed one or more variants of the umbrella species concept and identified 18 studies that represented empirical evaluations of the concept. Eight of those 18 represented empirical evaluations of the “area-demanding umbrella species” concept (Noss et al. 1996; Berger 1997; Martikainen et al. 1998; Andelman & Fagan 2000; Caro 2001, 2003; Suter et al. 2002)<sup>6</sup>; ten represented evaluations of the “site-selection umbrella species” concept (Murphy & Wilcox 1986; Ryti 1992; Launer & Murphy 1994; Kerr 1997; Andelman & Fagan 2000; Fleishman et al. 2000, 2001; Poiani et al. 2001; Rubinoff 2001; Bonn et al. 2002); and two studies represented evaluations of the “extended umbrella species” concept (Fleury et al. 1998; Watson et al. 2001). Their evaluation used the term “effectiveness” rather than “congruence” and they did not define the term “effectiveness”; however, their usage of the term “effectiveness” roughly corresponded with “congruence” as that term is generally applied.

Of the eight empirical evaluations of the “area-demanding umbrella species” concept, one study concluded that the concept was ineffective, two studies concluded the concept was either ineffective or had limited effectiveness, three studies concluded it had limited effectiveness, and the remaining two studies concluded it was effective for particular taxa, but not others. For example, a study that relied on the endangered capercaillie (*Tetrao urogallus*) was effective for other imperiled mountain birds, but was ineffective for avifauna that were not imperiled (Suter et al. 2002).

Of the 10 empirical evaluations of the “site-selection umbrella species” concept, two studies concluded that the concept was ineffective, one study concluded it was either ineffective or had limited effectiveness, three studies concluded it had limited effectiveness, and the remaining four studies concluded it was effective for particular taxa, but not others.

Based on their review, Roberge & Angelstam (2004) concluded that there was limited support for the “area-demanding umbrella species” concept, particularly if beneficiary species have more specialized habitat requirements than the umbrella species or if the spatial and habitat requirements of beneficiary species only partially overlap with those of the umbrella species. They concluded that the “site-selection umbrella species concept” was more useful. The effectiveness of both concepts increased when a study used multiple

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<sup>6</sup> Roberge & Angelstam (2004) produced eight separate evaluations by treating Andelman & Fagan (2000) as three separate studies, two of which were evaluations of the area-demanding umbrella species concept.

umbrella species and used systematic selection criteria (citing Fleishman et al. 2000, 2001). They concluded that the extended umbrella species concept had not been evaluated (but see discussion under **Section 5.2.2 Focal Species** for further discussion).

More importantly, Roberge & Angelstam (2004) concluded that only two of the 18 studies they evaluated (except Caro 2001, 2003) directly evaluated the basic assumption of the umbrella species concept: that the conservation measures directed at the umbrella species actually protects the intended beneficiaries.

### 5.2.2 FOCAL SPECIES

Studies that have tested the focal species concept can be grouped into two categories: studies that tested applications of the concept as originally envisioned by Lambeck (1997) and those that tested applications of a different conception of “focal species.” As described earlier, Lambeck (1997) developed the idea of “focal species” because of the limitations of the umbrella species concept due to its use of a single species to represent a pool of species. Instead of relying on a single species, he advocated using a suite of species that were selected to represent a pool of other species that use patches of habitat in a landscape and whose occurrence patterns (patterns of presence and absence) would be representative of occurrence patterns of a larger pool of species.

Caro (2010) argued that Lambeck’s conception of “focal species” was reasonably precise, and provided examples from Australia (Watson et al. 2001), Nova Scotia (King & Beazley 2005), and Belgium (Maes & Dyck 2005) to support his argument. The latter study (Maes & Dyck 2005) compared single-species (umbrella species) and multi-species (focal species) approaches and concluded that the multi-species (focal species) was more representative.

Roberge & Angelstam (2004) argued that Lambeck’s conception of “focal species” extended the umbrella species concept by proposing systematic criteria for identifying suites of species that have different sensitivities to the size and configuration of potential reserves in landscapes as well as to the compositional and functional attributes (i.e., habitat quality) of those areas.

Although Caro (2010) concluded that single species and multi-species approaches produced complementary information, he also wrote that the scientific consensus was that relying on multiple species (focal species) would produce results that are more representative of a larger suite of species than relying on a single species (citing Miller et al. 1999; Chase et al. 2000; Noss et al. 2002).

Nicholson et al. (2013) modeled expected extinctions to test how well different focal species combinations represented target taxa during reserve selection. As with other studies, their

results supported using more than one surrogate species to select reserves. In their study, three surrogate species provided the same outcome as the full set of ten beneficiary species. They concluded that the focal species were representative because their models indicated that the reserve system maximizing the persistence of the three focal species would also maximize the persistence of the ten beneficiary species. However, they cautioned that the success of their focal species to represent the beneficiaries might have been because of the relatively small number of beneficiary species (10), most of which could use all habitat types. They concluded that their results did not establish that focal species would be representative of larger suites of beneficiary species if the habitat requirements and distributions of the beneficiaries differed from the focal species. They raised two questions about the focal species selection process which remain unanswered. They are: (1) How many focal species, or what proportion of the total number of focal species are necessary to select to achieve good coverage of the beneficiary species? and (2) Given that their assessment of how well the focal species criteria worked was relative to only 10 beneficiary species, how large should the set of beneficiary species be?

Other investigators have reported that the focal species concept does not work. Lindenmayer et al. (2002) argued that the validity and practicality of the focal species concept were questionable because of the impracticality of collecting the data necessary to identify appropriate focal species. They argued instead for using multiple approaches centered on empirical data collection and possibly including single-species surrogates (umbrella species) and multi-species surrogates (focal species).

### 5.2.3 LANDSCAPE SPECIES

The validation of surrogacy assumptions for landscape species is fundamentally similar to those for umbrella and focal species because the criteria for selecting landscape species developed out of the surrogacy assumption testing for focal and umbrella species; therefore the results are not repeated here.

### 5.2.4 BIODIVERSITY INDICATORS

The assumption that biodiversity indicators can function as surrogate species, representing overall biodiversity or biodiversity in other specified taxa, has been tested many times over many decades. Caro (2010) summarizes the results prior to 2010 as promising, especially at larger geographic scales. Results since then generally support Caro's summary but are mixed. Two consistent recommendations for biodiversity indicators supported by the literature are: (1) it is beneficial to use more than one surrogate species and (2) defined boundaries should be used – systems or geographic scales should not be mixed (Lewandowski et al. 2010; Mellin et al. 2011; Larsen et al. 2012; Eglington et al. 2012; Westgate et al. 2014).

A number of investigators have focused on the strength and significance of correlation between surrogates selected as indicators of biodiversity and beneficiary species, communities, ecosystems, regions, or other components of biodiversity (Lawler et al. 2003; Xu et al. 2008; Heino 2010). The term used to refer to the strength of these correlations is “cross-taxon congruence.” To be effective in guiding conservation planning, most authors argue that there should be “strong congruence” between biodiversity indicators and the beneficiary species or other components of biodiversity (for example, correlations among taxonomic groups should preferably be positive and high). So far, many studies have reported high variability in cross-taxon congruence with substantial variation in which taxa, where, and how correlations are present (Caro 2010; Heino 2010; Trindade-Filho & Loyola 2011; Fattorini et al. 2011; Januchowski-Hartley et al. 2011; Eglinton et al. 2012; Westgate et al. 2014).

Heink and Kowarik (2010b) reviewed 56 studies for common approaches for selecting indicators for both ecology and environmental policy; the studies were also reviewed to determine against which criteria biodiversity indicators were scientifically tested. They found that only a few of the criteria which biodiversity indicators are expected to represent were empirically tested, leaving a lot of room to improve how indicators are used in ecology and environmental policy. They suggest taking two steps prior to selecting indicators. First, establish criteria to help determine whether the indicators’ reflect the aspects of the biodiversity that are of interest. Second, use scientifically sound methods to test whether the indicators meet the criteria being used to select them.

Combining the summary from work by Caro (2010) and Heink and Kowark (2010b) the following patterns, which become more complex and less universal at finer scales, appear to be emerging from studies reported in the literature:

- Species rich groups perform better as biodiversity indicators than groups with low numbers of species (Larsen et al. 2009, 2012; Mellin et al. 2011);
- More diverse surrogate groups of biodiversity indicators (i.e., species from different classes of organisms) perform better than less diverse surrogate groups (Larsen et al. 2009, 2012; Mellin et al. 2011; Fattorini et al. 2011);
- Different groups perform better in different areas; in other words, a taxa group shown to be a good surrogate for biodiversity in one region is unlikely to be a good surrogate in another region (Lewandowski et al. 2010; Mellin et al. 2011; Eglinton et al. 2012; Westgate et al. 2014);
- The most typical measure used in studies, species richness, may not be the best indicator of biodiversity (Fattorini et al. 2011; Eglinton et al. 2012); and
- There are circumstances/regions/systems where environmental or physical variables do a better job of assessing potential biodiversity than surrogate species; the optimal



solution often appears to be a combination of environmental data and biodiversity indicators (usually referred to as cross-taxon surrogates in this context) (Ferrier & Watson 1997; Rodrigues & Brooks 2007).

There are mixed results from studies conducted to test the biodiversity indicator surrogacy assumption, making it difficult to generalize results. A number of meta-analyses conducted and published within the past 5 years report limited success for biodiversity indicators as surrogates. Eglinton et al. (2012) conducted a meta-analysis of avian biodiversity indicators and concluded that bird diversity only weakly reflected other taxa and had limited utility. Westgate et al. (2014) conducted a meta-analysis of many sources and found that studies of cross-taxon congruence rarely gave consistent results. They found that species richness congruence appeared highest at extreme spatial scales and in geographies closest to the equator, while congruence in species composition appeared highest at large extents and grain sizes. A meta-analysis of biodiversity indicators in marine systems found that higher taxa were valuable surrogates only when they reflected species-level patterns of beta diversity (Mellin et al. 2011). Mellin et al. (2010) also concluded that the surrogates worked best in areas with low biological and functional diversity and over larger areas. Heino (2010) conducted a meta-analysis of freshwater biodiversity indicators and found that they did not work well as surrogates.

Lewandowski et al. (2010) completed a study that compared complementarity and hotspot approaches. They reported greater success with complementarity approaches, where surrogate species were 25% more likely to represent biodiversity as compared with hotspot approaches. Irrespective of approach, biome, and extent of study, surrogate taxon significantly influenced how well the surrogate performed.

In another study by Myšák & Horsák (2014), vascular plants were found to be good indicators of snail biodiversity in two distinct habitat types (treeless fens and forests). However, environmental gradients accounted for all significant positive correlations of species richness and therefore the authors concluded that the indicators had limited usefulness for conservation planning. During an evaluation of potential indicator species (or possibly umbrella species), Wesner & Belk (2012) found that fish diversity was higher at sites with potential indicators than sites without, indicating that conservation aimed at any of these species was likely to benefit other taxa.

In contrast, during a review of higher-taxon surrogates, Bevilacqua et al. (2012) found the surrogates performed no better than random assemblages of species. They suggested that species be grouped by more informative categories than higher level taxon categories.

In summary, studies testing the assumptions of biodiversity indicators as surrogates for overall biodiversity or biodiversity in other specified taxa report a mix of results, but most suggest limitations on their use.

#### 5.2.5 ENVIRONMENTAL SURROGATES

Evaluations of studies using environmental surrogates (i.e., surrogates other than species) by themselves, and used in conjunction with surrogate species, report mixed results. Some studies indicate that environmental surrogates can be effective on their own or can significantly enhance conservation planning when used in conjunction with surrogate species (Ferrier & Watson 1997; Carmel & Stroller-Cavari 2006; Carvalho et al. 2011; Januchowski-Hartley et al. 2011; James et al. 2012; Schindler et al. 2013; Lindenmayer et al. 2014; Di Minin & Moilanen 2014). Other studies indicate that habitat or other environmental measures are ineffective surrogates for species diversity (Araújo et al. 2001; Rodrigues & Brooks 2007). The overall effectiveness of using habitat or other physical conditions as surrogates for conservation planning is mixed, depending on scale, region and target taxa (Ferrier & Watson 1997; Grantham et al. 2010; McArthur et al. 2010; Carvalho et al. 2011; Walz 2011; Schindler et al. 2013; Lindenmayer et al. 2014).

### 5.3 MANAGEMENT AND ENVIRONMENTAL INDICATORS

There have been many studies assessing the usefulness of management or environmental indicators as surrogates. The biggest drawback to using indicator species as surrogates for other species' response is that each species may respond differently to the changing conditions. Species have unique environmental niches and may have different abilities to cope with or adapt to environmental changes, so monitoring one or a few species may not indicate the effects of the change on all the other species (Caro et al. 2005). When monitoring population trends, signals tied to the stressor of interest must be distinguishable from unrelated variation (Carignan & Villard 2002).

Morrison et al. (1992) critiqued the use of indicator species because of the unlikelihood of one species precisely reflecting the response of another species or group of species. For this assumption to be validated, the ability of the beneficiary and indicator species to tolerate or adapt to the particular environmental or management condition of interest needs to be the same. There are some theoretical reasons why this may not be a sound assumption and is rarely validated (Sætersdal & Gjerde 2011), not the least of which is that validating the representativeness of the indicator species to other species response of the same condition or change of interest is very difficult. One possible way of evaluating this was described in Butler et al. (2012), where they evaluated selected indicator species as compared with various past and modeled future land use data to determine how effective these indicators were at representing other species. They selected a suite of species that included species that exploited all resources used by the wider community and had the most specialized

resource requirements possible. Using this process to select indicator species, they compared the representativeness of a suite of 12 bird species with the niche space occupied by a different suite of 19 bird species; both were used to represent 62 beneficiary species. They concluded that the response of the 12 species to land-use changes was strongly correlated with the response of the beneficiary species, that their population trends closely matched the population trend of the beneficiary species, and that the focal species responded to land-use changes faster than the beneficiary community, so they would act as an early-warning indicator. They also concluded that the suite of 12 species selected based on resource requirements were better at representing the beneficiary species than the suite of 19 species that were currently being used and which had been selected via a different methodology.

*It is important to recognize the critical role monitoring plays in validating assumptions of surrogacy for management and environmental indicators as well as many other types of surrogates.*

Validating the surrogacy assumption of management or environmental conditions indicators requires extensive monitoring under multiple conditions and over time. For example, determining how well the surrogate species represent other species' responses to changing conditions of interest necessitates extensive monitoring of species and detailed environmental factors. Thompson (2006) suggests designing multi-faceted monitoring programs that allow managers to (1) respond to unexpected changes caused by management and (2) allow agencies to determine when a significant change has occurred and to predict the persistence of species. An abundance of literature describing the strengths and weaknesses of various monitoring designs and programs exists and is beyond the scope of this document.

To test the relationship between environmental indicator surrogates and beneficiary species, Banks et al. (2014) used life history traits and modeled the response of four proposed surrogate species to an environmental toxicant and compared the surrogate response with the response of the target species. The results indicated that, based on their life history models, none of the four proposed surrogate species were effective indicators for the beneficiary species.

Another assumption that needs validation is that the abundance of one species can serve as a surrogate, or indicator, for another species. This is the premise behind many uses of

management and indicator species but rarely tested. Cushman et al. (2010) tested this assumption in birds, using presence-absence and relative abundance as well as a priori and empirical groupings. Study results indicated stronger patterns of surrogacy with empirical groupings as compared with the a priori groupings; however, little variance was explained. Their results indicated that for the 55 birds studied here, there was little surrogacy value for either presence-absence or abundance.

While some uses of environmental and management indicators are well validated (e.g., various uses of freshwater IBIs), others have not been validated at all (Lindenmayer & Likens 2011; Ruaro & Gubiana 2013; Banks et al. 2014). This is particularly true for some environmental indicators, where a particular species was chosen many decades ago to serve as a surrogate for a large taxa group and no or few validating studies of the actual representation has occurred. For example, honeybees have been used as the standard for toxin exposure for almost all insects. However, the few studies that have examined this



**Figure 5. The boreal owl (*Aegolius funereus*) is an indicator species for forest ecosystems in Europe and North America.** It meets the criteria of a an indicator (Butler et al. 2012) as it serves as an early warning for detrimental environmental changes, it responds to the changes in a predictable way, and it reflects the status of wider biodiversity.

have not validated that assumption (Banks et al. 2014). Furthermore, the few studies that have examined this assumption have almost all used modeling to address the question, as opposed to direct tests on the species of interest.

In an effort to validate the correlation between the potential indicator and the conservation element of interest, Shanley et al. (2013) examined whether the northern flying squirrel would be a valuable indicator for late seral stage forest in Alaska. Their results suggested that the squirrel's association with particular habitat characteristics of interest (i.e., particular composition and structural features) made it a valuable surrogate for assessing forest management, both in Alaska and in other parts of the species range.

Ficetola et al. (2007) compared the response of a suite of management indicator species, which they referred to as focal species and included mammals, birds, and reptiles considered most sensitive to disturbance from human recreation. They compared these indicator species to a larger suite of beneficiary species from the same taxonomic groups and reported that the responses were only weakly correlated. They concluded that the differences between the responses of the indicator species and the responses of beneficiary species made representativeness of the indicator species questionable.

In another example, Bachand et al. (2014) evaluated several potential management indicator species to assess species' response to ecosystem recovery where deer density was reduced in both cut and uncut boreal forest. They evaluated abundance, percent cover, and indicator value, and compared single versus multiple species surrogates. Their results indicated that moths and plants were better indicators than other groups and that birds, bees and carabid beetles were not good indicators for this purpose. Similar to other studies, species combinations were better surrogates than single species.

In a different part of the world, Hoare et al. (2012, 2013) evaluated the utility of forest birds as potential management indicators for assessing changes due to exotic mammal control. While they found correlations among sets of birds, **they also discovered that the correlations were not accurately predicted a priori, leading them to conclude that correlations needed to be identified before selecting an appropriate indicator species or set of species.**

Tulloch et al. (2013) describe a method for selecting management indicator species that provides better power for detecting trends and at lower cost. Tulloch states, "decisions about which species to monitor from a suite of different species being managed are hindered by natural variability in populations and uncertainty in several factors: the ability of the monitoring to detect a change, the likelihood of the management action being successful for a species, and how representative species are of one another." They present a method that uses an index specifically developed for selecting indicators and accounts for the likelihood of successfully detecting a real trend resulting from a management action. They conclude

that poorly designed monitoring programs can be more expensive and inefficiently use conservation resources if they disregard uncertainty, the likelihood of detecting change, and the complementarity between species.

Freshwater IBI, a multimetric concept to assess the biological conditions in aquatic ecosystems, has been used as a tool to assess the effect of multiple stressors in aquatic environments for decades. IBIs were first introduced in 1981 by James Karr in a paper titled *Assessment of biotic integrity using fish communities* (Karr 1981). A recent study (Ruaro & Gubiana 2013) looked at the contribution of the IBI to the management of aquatic ecosystems by analyzing papers citing Karr (1981) and published between 1981 and 2011. Their results indicate that freshwater IBIs have been validated a number of ways and suggest that the ideas proposed by Karr have contributed to the conservation of aquatic ecosystems. They point out a few issues that need to be addressed to increase the robustness of the index, including the development of criteria for choosing different metrics and defining reference conditions.

#### 5.4 FLAGSHIP SPECIES

The primary assumption associated with the use of flagship species is that conservation efforts directed towards flagship species will result in increased awareness or support from a target audience for conservation of that flagship species and its habitat (Caro 2010). **With the inclusion of marketing methods and more detailed assessments of target audiences (Abeyta et al. 2011; Veríssimo et al. 2014a, 2014b; Bennett et al. 2015; Nekaris et al. 2015) rapid changes are occurring in the selection methodologies for flagship species. These newer methods inherently allow for testing of the assumption that flagships are increasing public support within target audiences.** Newer studies are also explicitly testing whether the use of flagship species is improving support for and awareness of other species.

Veríssimo et al. (2011, 2014a) considered how environmental economics and social marketing for target audiences shaped those audiences' attitudes and interactions with potential flagship species. Their results suggest that use of an evidence-based and audience driven selection framework results in the selection of more effective flagship species and that before working with potential target audiences, conservationists should specify the purpose of a marketing campaign. They encourage conservationists to monitor the success of their campaigns and to feed this information back into the marketing process. To determine whether using a flagship species is the best way to reach the desired outcome they also recommended conducting a return on investment analysis to decide whether it makes more fiscal sense to use a well-known species or to raise the awareness of a lesser-known flagship species.

Barua et al. (2011) found that the symbolic nature of flagships could invoke different reactions from different audiences and suggested that these reactions be understood before selecting a flagship. For example, Barua et al. (2010) discussed how the selection of a wolf, a tiger, or an elephant as a flagship species could result in strong support for their conservation from one group and an equally strong negative reaction from another group. Other studies indicate that flagship species may not actually increase support or funding for the conservation target (Tisdell & Wilson 2003) or protect other species in the ecosystem (Zacharias & Roff 2001) and conclude that careful monitoring is required to ensure a flagship species is functioning as expected.

While there is greater effort being conducted to validate the assumption that flagship species result in increased public awareness, support or funding for the flagship, there is still limited research on whether conservation efforts directed at flagship species result in benefits to other (beneficiary) species.

Bennett et al. (2015) used a project prioritization process developed in New Zealand and theoretically allocated additional private funding based on several scenarios to 10 or 22 existing flagship species. Random allocation of the additional private funds resulted in benefits to five additional species. Allocating this funding to projects that benefited the 10 primary flagship species benefited six additional species – only one more than the random allocation. Allocating this funding to the larger suite of 22 flagship species also resulted in gains of up to six species, and generally greater biodiversity gains across budgets. The scenario where the private funding went to general biodiversity goals resulted in the greatest biodiversity gains with benefits up to 13 additional species and greater phylogenetic diversity of species that benefited.

Smith et al. (2012) analyzed the flagship campaigns for 59 organizations that used 80 flagship mammal species to assess the results of fundraising from those campaigns. They found that 61% of the raised funds for flagship species went towards the conservation of only the flagship species. This indicates that flagships may be serving partially as surrogates to raise funds for other species, but they are not currently doing that very well.

## 6 ASSESSING USE AS A CONSERVATION TOOL

As discussed in **Section 5**, any conservation effort that relies on a surrogate makes two fundamental assumptions: (1) the surrogate is an appropriate representative of a larger group of species, biotic communities, or ecosystems and (2) conservation efforts developed for the surrogate will benefit other species, biotic communities, or ecosystems. **Section 5** focused on the first of these assumptions, which involves near-term monitoring and research and does not typically evaluate whether the use of surrogate species actually helps achieve a conservation goal or objective. This section focuses on the second assumption; testing this second assumption requires long-term monitoring designed to assess whether conservation objectives are being achieved and identifying ways to improve management and conservation priorities. While **Section 5** was organized around different surrogate species concepts, this section is organized around different uses of those concepts.

As described in **Section 5**, a small number of studies have evaluated the assumption of surrogacy for any given surrogacy approach and many of those evaluations have been conceptual rather than empirical. Even fewer studies have evaluated actual conservation efforts to determine whether or to what degree beneficiary species, communities, or ecosystems have benefitted from decisions and management actions designed to benefit a surrogate. Part of the problem is that this kind of evaluation requires long-term monitoring of conservation efforts, which rarely occurs. Another problem is that there is disagreement about the appropriate methods that should be used to evaluate conservation efforts (Gardner 2010; Bottrill & Pressey 2012). As a result, different investigators use different evaluation criteria, which make it difficult to compare evaluations.

### 6.1 LANDSCAPE DESIGN AND RESERVE SELECTION

To test a regional bird conservation plan developed by Partners in Flight, Alexander et al. (2007) compared the abundance of a suite of 12 bird species that had been selected as surrogate species in untreated stands (referred to as focal species in the paper) to their abundance in treated stands where the “treatment” consisted of purposefully-reduced shrub cover to lower the risk of fire. The surrogate species were originally selected to represent particular elements of the landscape and the plan resulted in a variety of conservation actions. They evaluated whether the surrogate species increased or decreased in abundance as a result of the treatments. Over a two-year period, two of twelve oak woodland and chaparral surrogate species were more abundant at treated units; no species were consistently less abundant at treated units. Although this represents a short-term study, these results suggest small-scale (7–42 ha) treatments were consistent with the objectives identified in the Partners in Flight regional conservation plan because they benefited species associated with edges, but did not have negative effects on shrub-associated species. Therefore, implementing one of the management actions of the conservation plan for this



landscape did benefit the surrogate species, but effects on beneficiary species were not tested.

One of the most complete examples of assessing whether a designed system benefited species other than the surrogate species is a study of marine reserves that Olds et al. (2014) conducted in the Solomon Islands. These reserves were designed on the basis of local ecological knowledge to conserve bumphead parrotfish (*Bolbometopon muricatum*) and to protect food security and ecosystem functioning. Olds et al. (2014) examined the value of this species as a conservation surrogate and assessed the importance of seascape connectivity among reefs, mangroves, and seagrass to marine reserve performance. They evaluated a suite of variables and attributes of the reserves, but generally concluded that reserves designed for bumphead parrotfish benefitted at least 17 other fish species based on their abundance patterns.

In a test of both potential surrogacy of connectivity and current surrogacy of existing protected areas, Epps et al. (2011) evaluated whether the African elephant (*Loxodonta africana*) would serve as an appropriate surrogate species for other large mammals in a potential corridor between two major protected area complexes, as well as whether current reserves designed for elephants protected other mammals. They found that elephant presence was highly positively correlated with the richness of large mammals. Outside of protected areas, both mammal richness and elephant presence were negatively correlated with human population density and distance from water. This is a particularly informative study because the elephant is a widely used flagship species as well and shows that flagship species can also serve in a role as umbrella species for a diversity of other species.

In a test of Lambeck's (1997) focal species approach for a reserve design application Nicholson et al. (2013) found that selecting areas based on the needs of the most restricted species (as suggested by Lambeck (1997)) did result in a reserve design that covered the rest of the species in the landscape. Caro (2003) found that using umbrella species as a means of encompassing populations of co-occurring species at local scales was effective because populations of beneficiary species were still numerous in most reserves in East Africa, 50 years after their creation; most reserves were large and could encompass substantial populations of beneficiary species. He concluded that the local-scale umbrella-species concept should not be discarded despite its conceptual difficulties.

## 6.2 BIODIVERSITY INDICATORS

Few studies have evaluated the use of biodiversity indicators to achieve a conservation objective other than reserve selection (presented separately below). As discussed in Caro (2010), the literature regarding the most appropriate biodiversity indicators for global and continental scales is extensive by itself. There has been much discussion and debate about the utility and results of using rare species, endemic species, unique species, total species

richness, and other measures of biodiversity hotspots at global and continental scales. Each approach has its benefits and drawbacks and, again, the selection of the appropriate biodiversity indicator at that scale is dependent on the conservation goal.

A meta-analysis of biodiversity indicators (referred to as umbrella species) examined species richness and abundance of co-occurring species (Branton & Richardson 2010). They found that richness and abundance were consistently higher in sites where biodiversity indicators were used for conservation planning, than when they were not. Birds were generally better biodiversity indicators than mammals, with smaller birds generally better than bigger birds and omnivores better than herbivores or carnivores. Larger body size and/or home range did not translate to better biodiversity indicators. Similarly, taxonomic similarity, resource specialization, or trophic level did not correlate with better biodiversity indicators. As discussed by Branton & Richardson (2010), because most studies examining this use of surrogate species have focused only on birds, plants and a few groups of insects, it is an incomplete assessment.

### 6.3 MANAGEMENT AND ENVIRONMENTAL INDICATORS

There is a substantial literature, separate from surrogate species that evaluates and reviews monitoring programs and techniques, including the use of management and environmental indicators as surrogate species. This literature is not reviewed here but some recent reviews provide summaries of the current state of the research (e.g., Caro 2010; Noon et al. 2012; Ruaro & Gubiana 2013), as well as many articles in the journal *Ecological Indicators* (available at <http://www.journals.elsevier.com/ecological-indicators/>).

### 6.4 FLAGSHIP SPECIES

Gauging public and stakeholder support for the management of beneficiary species as a result of the use of a flagship species is the core of efficacy testing of flagship species. To determine efficacy, monitoring efforts need to track the public's level of support for management of beneficiary species, both with and without using the flagship. To date, the impacts of flagship species on public attitudes and the ability to deliver strategic conservation goals have not been well evaluated (Barua et al. 2011).

Veríssimo et al. (2014a) evaluated the effectiveness of an existing flagship campaign with the Araripe manakin (*Antilophia bokermanni*) in Brazil. Almost all respondents recognized the species only by its more recent common name and knew that it was threatened. The study's results indicated that the flagship campaign was accomplishing its conservation awareness goal; a third of respondents recognized the importance of springs and reduced habitat degradation to its conservation and some respondents recognized local human communities as important in conservation. The study did not evaluate whether conservation outcomes or conservation behaviors were changing as a result of increased awareness of the

flagship. The increased conservation awareness is a strong indication that this flagship species is working as intended.

Home et al. (2009) evaluated flagship species from 14 organizations and measured whether the flagships were able to motivate support. A charismatic species and a less charismatic species were selected as treatments in a quantitative experiment with 900 respondents. The results indicated that both charismatic and less appealing species had the ability to change public preferences for habitats supporting biodiversity in urban landscapes.

Barua et al. (2011) recommended ways to evaluate and improve the efficacy of flagship species. They acknowledged that the effectiveness of a flagship is linked to the charisma it provokes, a trait that is specific to audiences and cultures. They suggested that the absence of a clearly defined audience in addition to a lack of data on species' charisma as inputs to the selection process, could reduce the flagship's functionality at increasing public support.

Evaluating whether a flagship is supporting a conservation objective (usually relating to conservation awareness or public support) requires evaluating the effect of the flagship species on the target audience with respect to the specific conservation objective for which it was selected. Historically, flagship species were not selected for a specific audience nor with a defined conservation objective, which made evaluating their efficacy difficult. More recent studies are correcting this and should result in an improved understanding of the value and performance of flagship species as conservation tools in the future.

## 6.5 LONG-TERM MONITORING AND EFFICACY TESTING

Monitoring is critical for determining whether conservation plans are meeting their goals, yet comprehensive monitoring programs to assess the efficacy of conservation plans to meet their stated objectives are often lacking. This section describes potential monitoring approaches that could be developed to evaluate the success or failure of surrogate species to achieve conservation goals; the examples described would need to be altered to meet the specific needs associated with evaluating the efficacy of the surrogates.

Monitoring activities often occur in an ad hoc and piecemeal fashion that lack the ability to assess the effectiveness of the conservation plan as a whole – let alone the benefits from a specific part of the plan over a long period of time (Franklin et al. 2011). It is especially important to measure environmental drivers and stressors in addition to target species and community parameters, allowing managers to determine whether the plan's objectives are being met and at the same time update their knowledge of the system, ultimately informing adaptive management decisions. Monitoring programs must be designed to have adequate power to determine the status and trend of the variable of interest.

Most of the studies described earlier in this section provide some assessment of whether a surrogate approach is achieving a conservation objective; however, few of these studies

really assess whether the surrogate approach helps achieve the conservation plan over the long term by increasing the viability of the beneficiary species over time. The only way to test this is with a comprehensive monitoring program over an adequate number of years.

There are multiple approaches for accomplishing the type of monitoring needed to assess whether a conservation plan is achieving its intended goal(s) and whether the surrogate species are helping accomplish that goal or not. The following sections describe some of the different monitoring approaches that could be used to gather the necessary data.

### 6.5.1 MULTI-SPECIES MONITORING

Franklin et al. (2011) tested a multi-species monitoring framework modified from Atkinson et al. (2004), using San Diego's Multiple Species Conservation Program in southern California as a case study. The framework included the following interconnected steps: (1) prioritization of species and communities (biodiversity elements) for monitoring, based on risk and representation; (2) development of conceptual models that identified specific conservation objectives, critical monitoring variables, important threats, and management responses; and (3) use of existing data to assess key components of spatial and temporal variability in some of the monitoring variables. The following steps are essential to this type of monitoring program.

1. Identify habitat conservation plan goals – clear and concise conservation goals leading to specific objectives are essential precursors to any successful ecological monitoring program.
2. Prioritize covered species for monitoring – in an attempt to logically allocate scarce resources to the monitoring required, they adapted a risk-based species prioritization scheme with the intention of protecting multiple species
3. Prioritize natural communities for monitoring – possible criteria include areal extent, representativeness, fragmentation, and endangerment.
4. Identify goals and objectives for priority monitoring elements – identifying and refining conservation objectives for prioritized species and communities is an important precursor to designing the monitoring program (i.e., where, when, and how to monitor the element of concern).
5. Develop conceptual models – the development of conceptual models has been identified as a critical tool for conservation plans (e.g., Atkinson et al. 2004). These models can be narratives or diagrams, tables or matrices, but they link causes (stressors, threats, drivers) with effects on the state of the environment or biotic responses. The model should include explicit links to decision-making or management actions.
6. Design monitoring program

Another example of multi-species monitoring is presented by Manley et al. (2004, 2006). **They present a multiple-species monitoring approach using simulations compared to field sampling results to estimate species detections. They suggest their approach as an effective and feasible alternative to large-scale monitoring programs by targeting the most basic of population data for a large number and breadth of species. To determine whether the populations of these species remain viable, the monitoring would have to continue over a long time period.** Through simulations compared to field sampling results, they estimated that adequate detections would be obtained for 76% of the 465 vertebrate species (excluding fish), including 83% of all birds, 76% of all mammals, 65% of all reptiles, and 44% of all amphibians. Detection adequacy varied among life-history and ecological groups, but >50% of the species were adequately detected in every group with the exception of three groups: rare species, endemic species, and species of concern (33%, 24%, and 47% of associated species adequately detected, respectively).

Barrows et al. (2005) propose a framework for uniting single species and ecosystem approaches to address the monitoring needs of multiple-species conservation programs. Their approach employs primary data collection to build conceptual and quantitative models, habitat condition assessments, and species-specific surveys that allow linking species population trajectories with community or ecosystem processes and conditions. They illustrate implementing this method with seven threatened species, including plants, invertebrates, reptiles and mammals in the Coachella Valley in the Colorado Desert of southern California. The result was a database that spanned trophic relationships, tracked potential stressors, allowed analysis of inter-specific patterns in abundance and distribution, and evaluated the effect of the various drivers on the abundance of the covered species. This was an intensive effort, however, and only produced information on the targeted species.

These types of long-term, multi-species monitoring efforts require adequate institutional and funding support. Time committed to carefully developing a monitoring program will assure that limited resources are put toward the highest priority conservation elements and that monitoring will inform managers about the status of those elements.

### 6.5.2 ASSESSMENT OF ECOLOGICAL INTEGRITY

Another approach to assessing the efficacy of a conservation program is to assess the functionality or overall health of a landscape, which requires the evaluation of patterns and processes. This is usually referred to as assessing the functional or ecological integrity of an ecosystem or landscape (Parrish et al. 2003). Assessment of ecological integrity or functionality is still an evolving area of research (Poiani et al. 2000). Evaluation of ecological patterns and processes represents a major challenge for conservation scientists and practitioners. The difficulty in assessing the ecological integrity of a landscape is

translating functional attributes into effective, useful, and measurable specifics for planning, monitoring, and assessment (Poiani et al. 2000).

Parrish et al. (2003) recommend four metrics to use to assess ecological integrity of a landscape: (1) identify a limited number of conservation targets, (2) identify key ecological attributes for these targets, (3) identify an acceptable range of variation for each attribute as measured by properly selected indicators, and (4) rate target status based on whether or not the target's key attributes are within their acceptable ranges of variation. Examples of the approach were not presented and the authors agreed that many see its implementation as daunting.

Using another approach, a set of metrics to monitor for ecological integrity can be developed from (Fischer & Lindenmayer 2007). Adapting their list of comprehensive considerations for what to monitor in a landscape results in two lists of elements to monitor.

#### Pattern-oriented elements

1. large and structurally complex patches of native vegetation
2. a matrix that is structurally similar to native vegetation
3. buffers around sensitive areas
4. corridors and stepping stones
5. landscape heterogeneity and environmental gradients

#### Species-oriented elements

1. key species interactions and functional diversity, protect:
  - a. important ecosystem processes
  - b. characteristic ecosystem structure and feedbacks
2. appropriate disturbance regimes, protect:
  - a. characteristic vegetation structure
  - b. characteristic spatial and temporal variability in vegetation patterns
3. species of particular concern, protect the:
  - a. survival of rare or threatened species
4. aggressive, over-abundant and invasive species, indicate:
  - a. competition and predation by undesirable species which could negatively affect desirable species, and;
  - b. characteristic species composition
5. ecosystem-specific threatening processes, may indicate:
  - a. problems that affect biodiversity but are not directly related to landscape modification

This is an area of ongoing research where innovative ways of monitoring the ecological integrity are being developed and tested. Tying this monitoring to the adaptive management of a conservation plan is yet another step.

### 6.5.3 INDEX OF BIOLOGICAL INTEGRITY

A third way of monitoring the efficacy of a conservation program is to use suites of indicator species, such as the IBIs discussed earlier. While the initial and best validated uses are in freshwater systems (Whittier et al. 2007; Ruaro & Gubiana 2013), there have been a number of IBIs developed for terrestrial systems (e.g., O'Connell et al. 2000; Karr & Kimberling 2003; Noson & Hutto 2005).

O'Connell et al. (2000) used suites of birds to evaluate the biological integrity of forested landscapes, asserting that the resulting IBIs could be used to assess the overall structural, functional, and compositional condition of ecosystems. They asserted that bird communities can indicate high integrity if dominated by guilds dependent on native system attributes, where specialists are well represented relative to generalists. The resulting Bird Community Index (BCI) is based on response guilds, which are groups of species that require similar habitat, food, or other elements for survival. They recommend that information from the BCI be combined with that from additional indicators for a robust assessment of an area.



**Figure 6. A Fish Index of Biotic Diversity is a measure to monitor stream health in New Jersey.** While one or more surrogate species are chosen to represent functional ecosystems in planning, to test whether that approach is effective can require a robust index (Karr 2006) including taxa richness, indicator taxa, health of individual organisms, and assessment of biological processes.

## 7 CONCLUSIONS

Surrogate species have been defined as species which represent other species or aspects of the environment and are used to attain a conservation objective. Throughout the literature, one of the statements made by many authors is that the use of surrogate species is necessary. Managers cannot identify the habitat and resource needs of every species in a landscape; monitor environmental or management effects on every species; or directly monitor all of the workings, interactions and threats in the environment, so using surrogate species becomes inevitable even when it is not explicitly recognized.

Inconsistent use of the terms, concepts, and definitions of surrogate species has created challenges for evaluating their usefulness and improving their effectiveness in conservation planning. There is much confusion and misuse of surrogate species terms, even within the scientific literature. Any use of a surrogate term should be accompanied with a clear definition.

Surrogate species can be grouped into three categories, based on the conservation objective(s).

1. Surrogate species used to define conservation areas (including landscape design), include biodiversity indicators, umbrella species, focal species and landscape species. Application of these different surrogate species types are appropriate at different geographic scales and have different levels of effectiveness. The theory and processes associated with their use continue to evolve and become more robust and rigorous.
2. Surrogate species used to document the effects of management or environmental conditions include management indicators, environmental indicators, management umbrella species and cross-taxon indicators. These species have multiple applications throughout the adaptive management cycle.
3. Surrogate species used to gain public support are called flagship species. A flagship species could also be a species used under one of the other surrogate approaches, but the flagship, by definition, must improve public support, conservation awareness or fundraising that benefits other species.

**The type(s) of surrogate species selected and their selection criteria are directly related to the conservation goal and objectives trying to be achieved; therefore a critical first step is clearly defining the goals and objectives.** There is one criterion for selecting surrogate species that apply regardless of the conservation goal or surrogate type. It is that surrogates should be representative of the beneficiary species, communities and/or habitats so that actions taken to benefit the surrogate also benefit co-occurring species. Other selection criteria depend upon the conservation objective and the type of surrogate species.



There are many criteria selection processes that have been developed and described in the published literature and they continue to evolve as the science of using surrogate species matures. The assumptions made during any application should be explicitly stated and tested.

While there is still a significant amount of research needed on surrogate species to confirm the assumptions associated with their use; to develop guidelines for their use; and to identify better methods for their selection, there are examples of their viable and productive uses in conservation planning. Currently, the lack of published results on the use of surrogate species limits the understanding of their value or lack thereof. Few publications address the actual effectiveness of surrogate species (alone or in combination) for achieving specific conservation objectives. Even fewer studies describe what is learned by using surrogate species as part of an adaptive management framework. For example, are management strategies revised if unexpected results are obtained by monitoring surrogate effectiveness? While using surrogate species reduces the total number of species needing evaluation, substantial effort, including data collection, must occur to achieve a specific outcome and ensure the validity of their use.

Directed research and monitoring are two primary methods for evaluating the success of surrogate species and modifying their use based on evaluation results. Both directed research and monitoring data can be used for evaluating and updating assumptions, validating model predictions, and gauging progress toward conservation objectives. Long-term monitoring data provides the raw material for comparing expected conservation outcomes with actual conservation outcomes. By incorporating the feedback from research and monitoring the uncertainties in assumptions and predictions can be reduced, the models revised, and the overall credibility and effectiveness of a conservation strategy improved.

**The literature reports differing success among the types of surrogate species used.** The most successful applications of surrogate species share (1) explicit goals for their use, (2) a careful selection process using well-defined criteria for achieving the stated goals, and (3) well designed monitoring for testing the efficacy of the approach used. In contrast, the main impediments to using surrogate species successfully have been (1) confusing terminology, (2) unclear objectives, and (3) incorrect or ambiguous implementation. For surrogate species to be effective, the concepts, goals, methodologies and specific applications of the different types of surrogate species used need to be explicit, and their intended objectives clear and measurable.

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## APPENDIX B: ACRONYMS

BCI	Bird Community Index
BLM	Bureau of Land Management
FWS	US Fish & Wildlife Service
HEP	Habitat Evaluation Procedures
IBI	Index of Biological Integrity
IPAs	Important Plant Areas
IUCN	International Union for Conservation of Nature
LCC	Landscape Conservation Cooperatives
LMEs	Large Marine Ecosystems
LSA	Landscape Species Approach
MPAs	Marine Protected Areas
OFN	Ontario's Far North
SHC	Strategic Habitat Conservation
USFS	US Forest Service
USGS	US Geological Survey

## APPENDIX C: GLOSSARY

Abiotic surrogate	Surrogate that is based on a physical, chemical or hydrological (or other non-biological) characteristic
Adaptive management	Adaptive management is a systematic process for continually improving management policies and practices by learning from the outcomes of operational programs.
Background species	Species that benefits from the use and management of a surrogate approach
Beneficiary species	The co-occurring species that are likely to benefit from conservation activities directed at a surrogate species.
Biodiversity	The variety and variability of living organisms and the environments in which they occur.
Biodiversity indicator	Species that can be used to identify areas that are important for conservation or function as environmental or management indicators.
Biotic surrogate	Surrogate based on a biological component of an ecosystem (habitat, species, vegetation component)
Cinderella species	Esthetically pleasing and overlooked species that fulfill the criteria of flagships or umbrellas.
Conservation goal	A broad statement about the desired outcome of the conservation program.
Conservation objectives	A specific outcome that is desired and identified during conservation planning
Conservation planning	Any planning activity to identify goals, objectives and actions to benefit one or more components of biodiversity
Environmental indicator	A component or a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals.
Environmental conditions indicator	Species used as surrogates to monitor the effects of environmental conditions on ecological systems, including the species supported by those conditions.
Evaluation species	Those fish and wildlife resources in the planning area that are selected for impact analysis.
Flagship species	Those species that are used to reflect or engender public support for conservation efforts.
Focal species	Suite of species, each of which is used to define different spatial and compositional attributes that must be present in a landscape and their appropriate management regimes.
Indicator species	An organism whose characteristics (e.g., presence or

	absence, population density, dispersion, reproductive success) are used as an index of attributes too difficult, inconvenient, or expensive to measure for other species or environmental conditions of interest.
Keystone species	A species whose population is the keystone of the community's structure, whereby the integrity and stability of the community are determined by its activities and abundance.
Landscape conservation	A conservation approach designed to focus on land and water problems at an appropriate geographic scale, regardless of political and jurisdictional boundaries.
Landscape design	A partnership-driven process and a product that results in a science-based, spatially-explicit representation of the desired future condition of that landscape needed to meet population objectives.
Landscape objective	An endpoint (or an achievable outcome) for conservation that is at the scale of the landscape.
Landscape species	Those species that use large, diverse areas (either species with large home ranges or species with wide distributions) and often have significant impacts on the structure and function of natural ecosystems.
Management indicator	Species used to track the effects of management actions on the species potentially affected by the management.
Marine Protection Areas	Marine, coastal, or estuarine areas (including intertidal areas and bays) where natural or cultural resources are given greater protection than the surrounding waters.
Monitoring	The process of evaluating results in order to assess the adequacy of the mitigation and conservation strategies or activities and to provide information to direct the adaptive management program.
Strategic Habitat Conservation	A science-based framework founded on an adaptive, iterative process of biological planning, conservation design, conservation delivery, monitoring, and research.
Surrogate species	Species used to represent other species (or, less commonly, aspects of the environment) to attain a conservation objective.
Target species	Species that is the target of a conservation plan, objective or action
Umbrella species	The presence of a particular species (the umbrella species) in a geographical area indicates that other species (the beneficiary species) will also be present.

## APPENDIX D: LITERATURE REVIEW SUMMARY

### Appendix D1: Complete List of Literature Consulted.....D-2

This reference list includes papers consulted during the development of this document. This list does include papers not cited in the document but are part of the broader scientific literature on surrogate species and associated conservation planning.

### Appendix D2: Summary of Review Papers.....D-36

This reference list includes major review papers after 2010 that critique, summarize, or perform meta-analysis on a particular aspect of surrogate species. A tabular summary is also provided that summarizes the major conclusions of each paper.

### Appendix D3: Summary of Primary Research since 2010..... D-49

This reference list includes primary research on surrogate species from 2010 to early 2015. Research prior to 2010 is generally not included, as it was assumed *Conservation by Proxy* (Caro 2010) and other reviews prior to 2011 sufficiently summarize primary research prior to 2010. A tabular summary is also provided that summarizes the purpose, methods and conclusions of each paper.

## APPENDIX D1: COMPLETE LIST OF LITERATURE CONSULTED

This reference list includes papers consulted during the development of this document. This list includes not only those papers cited in the document but also those that are part of the broader scientific literature on surrogate species and associated conservation planning that informed the content here, even when not cited directly.

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## APPENDIX D2: SUMMARY OF REVIEW PAPERS

This reference list includes major review papers after 2010 that critique, summarize, or perform meta-analysis on a particular aspect of surrogate species. A tabular summary is also provided that summarizes the major conclusions of each paper.

- Arponen, A. 2012. Prioritizing species for conservation planning. *Biodiversity and Conservation* **21**:875–893.
- Barua, M. 2011. Mobilizing metaphors: the popular use of keystone, flagship and umbrella species concepts. *Biodiversity and Conservation* **20**:1427–1440.
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- Mellin, C., S. Delean, J. Caley, G. Edgar, M. Meekan, R. Pitcher, R. Przeslawski, A. Williams, and C. Bradshaw. 2011. Effectiveness of biological surrogates for predicting patterns of marine biodiversity: a global meta-analysis. *Plos One* **6**. Available from ://WOS:000291682300003.
- Murphy, D. D., P. S. Weiland, and K. W. Cummins. 2011. A critical assessment of the use of surrogate species in conservation planning in the Sacramento-San Joaquin Delta, California (USA). *Conservation Biology* **25**:873–878.
- Noon, B. R., L. L. Bailey, T. D. Sisk, and K. S. McKelvey. 2012. Efficient species-level monitoring at the landscape scale. *Conservation Biology* **26**:432–441.
- Ruaro, R., and E. Gubiana. 2013. A scientometric assessment of 30 years of the Index of Biotic Integrity in aquatic ecosystems: Applications and main flaws. *Ecological Indicators* **29**:105–110.
- Sætersdal, M., and I. Gjerde. 2011. Prioritising conservation areas using species surrogate measures: consistent with ecological theory? *Journal of Applied Ecology* **48**:1236–1240.
- Sarkar, S. 2014. Biodiversity and Systematic Conservation Planning for the Twenty-first Century: A Philosophical Perspective. *Conservation Science* **2**:1–11.
- Schultz, C. A., T. D. Sisk, B. R. Noon, and M. A. Nie. 2013. Wildlife conservation planning under the United States Forest Service’s 2012 planning rule. *Journal of Wildlife Management* **77**:428–444.
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- Williams, B., and F. Johnson. 2013. Confronting dynamics and uncertainty in optimal decision making for conservation. *Environmental Research Letters* **8**:025004.

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
Westgate et al. 2014. Global meta-analysis reveals low consistency of biodiversity congruence relationships. <i>Nature Communications</i> 5 (3899).	Biodiversity indicators	Assessed 'congruence' across taxon - using species accumulation index	These results show that although there is a broad congruence between taxa in general, congruence is not typically high, whereas there is a high degree of variation in congruence between studies.	Surrogates generally more effective at poles and equators
Brock & Atkinson. 2013. Selecting species as targets for conservation planning. In: Craighead L, Convis C, editors. <i>Conservation Planning: Shaping the Future</i> . Esri Press. p. 123–147.	All, but primarily landscape species	Reviewed all surrogate species but then focused on one example - landscape species in Montana	They discuss use of surrogate species as shortcuts for developing conservation plans intended to ensure long-term survival of native species and functioning ecosystems within a planning area. They provide a brief overview of the use of surrogate species in conservation planning and include some general suggestions for practitioners seeking to implement this approach. Finally, they propose a new framework for selecting species as targets for site-based conservation planning.	Detailed recommendations to improve use and outcome of landscape species.
Ruaro & Gubiana. 2013. A scientometric assessment of 30 years of the Index of Biotic Integrity in aquatic ecosystems: Applications and main flaws. <i>Ecological Indicators</i> 29:105–110.	IBIs (management and environmental indicator species)	Overall use of IBIs	The results suggest that the ideas proposed by Karr have contributed to the conservation of aquatic ecosystems. However, criteria for choosing different metrics, as well as the definition of reference conditions, are issues that need to be addressed in order to make the IBI a more robust index.	Still some issues to be resolved (comparison across the global/continents; definition of reference conditions; use in tropical systems) but overall contributing to understanding and conservation aquatic systems.
Schultz et al. 2013. Wildlife conservation planning under the United States Forest Service's 2012 planning rule. <i>Journal of Wildlife Management</i> 77:428–444.	Umbrella species	Monitoring	Coarse filter approaches alone will not be sufficient to ensure the viability of wildlife populations. A comprehensive wildlife assessment framework would include a combination of both coarse- and fine filter approaches. It would commit to monitoring at-risk and focal species using recent	USFS needs to commit to direction in 2012 rule and not let flexibility kill the potential.

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			advances in monitoring approaches. As required for adaptive management, monitoring would occur at multiple spatial scales and use pre-defined triggers to evaluate the consequences of management actions and to inform future management decisions.	
Williams & Johnson. 2013. Confronting dynamics and uncertainty in optimal decision making for conservation. <i>Environmental Research Letters</i> 8:025004.	Conservation planning	.	Examples of how the dynamic optimization problem can be framed for problems involving management of habitat for an imperiled species, conservation of a critically endangered population through captive breeding, control of invasive species, construction of biodiversity reserves, design of landscapes to increase habitat connectivity, and resource exploitation.	The process and system are not static.
Arponen. 2012. Prioritizing species for conservation planning. <i>Biodiversity and Conservation</i> 21:875–893.	Conservation planning	Species prioritization	Criteria used for species prioritization range from aesthetical to evolutionary considerations. Two main aspects of diversity are used as objectives: Maintenance of biodiversity pattern and maintenance of biodiversity process. There are two additional criteria typically used in species prioritization that serve for achieving the objectives: The species' need of protection and cost and effectiveness of conservation actions. But preserving evolutionary process versus current diversity pattern may turn out to be conflicting objectives, if pursued simultaneously.	Although many reasonable criteria and methods exist, species prioritization is hampered by uncertainties, most of which stem from the poor quality of data on what species exist, where they occur, and what are the costs and benefits of protecting them. Surrogate measures would be extremely useful but their performance is still largely unknown.
Bevilacqua et al. 2012. Taxonomic relatedness does not matter for species surrogacy in the assessment of community responses to environmental drivers. <i>Journal of</i>	Biodiversity indicators	Testing surrogacy assumption	They evaluated the effectiveness of higher taxa as species surrogates and reviewed the current literature on taxonomic sufficiency to check for any correlation between the effectiveness of higher taxa and the degree of	The findings cast doubt on static taxonomical groupings, legitimizing the use of alternative ways to aggregate species to maximize the use of

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
Applied Ecology 49:357–366.			species aggregation across different types of organisms.	species surrogacy.
Eglington et al. 2012. A meta-analysis of spatial relationships in species richness across taxa: birds as indicators of wider biodiversity in temperate regions. <i>Journal for Nature Conservation</i> 20:301–309.	Biodiversity indicators - birds and temperate regions only	Testing surrogacy assumption	Species richness in birds only weakly reflected species richness in other taxa, with 19% of the variation in total species richness in other taxa was explained by species richness in birds. Birds were more effective at reflecting cross-taxa species richness in study areas dominated by agricultural mosaics or mixtures of habitats; they were less effective in forests and grassland environments. Overall, birds were better at reflecting species richness in mammals than other taxa, and relationships were more effective at larger spatial scales.	There is a need to assess whether temporal change in bird populations and assemblages, as opposed to spatial variation, reflects change in other taxa and to identify elements for which birds could be the most effective surrogates.
Noon et al. 2012. Efficient species-level monitoring at the landscape scale. <i>Conservation Biology</i> 26:432–441.	Monitoring Indicators	Monitoring design	Even with the efficiencies gained when occupancy is the monitored state variable, the task of species-level monitoring remains daunting due to the large number of species. A small number of species should be monitored on the basis of specific management objectives, their functional role in an ecosystem, their sensitivity to environmental changes likely to occur in the area, or their conservation importance.	Use occupancy methods and models (presence/absence in sampled units) for monitoring management.
Sarkar. 2014. Biodiversity and Systematic Conservation Planning for the Twenty-first Century: A Philosophical Perspective. <i>Conservation Science</i> 2:1–11.	Biodiversity indicators and other surrogates for conservation areas	Selecting conservation areas	Various software tools have been developed for implementing algorithms to identify conservation area networks for the representation and persistence of biodiversity features. This paper reviews the development of these tools and evaluates the suitability of different algorithms for their solution. They also review some key issues associated with the use of these tools, such as computational	

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			efficiency, the effectiveness of taxa and abiotic parameters at choosing surrogates for biodiversity, the process of setting explicit targets of representation for biodiversity surrogates, and dealing with multiple criteria.	
Smith et al. 2012. Identifying Cinderella species: uncovering mammals with conservation flagship appeal. <i>Conservation Letters</i> 5:205–212.	Flagship species	Relationship to fundraising	Flagship species are used to fundraise for broader issues but alternative species with similar appeal to the target audience could exist. International NGO flagship campaigns that use threatened mammal species were evaluated and data were used to identify “Cinderella species,” which are aesthetically appealing but currently overlooked species. The 59 NGOs examined only used 80 flagship species and that 61% of their campaigns only raised funds for the species itself. Existing flagships are generally large and have forward-facing eyes and that there are 183 other threatened species with similar traits.	The current approach is overly limited but NGOs could overcome this by adopting some of these Cinderella species as new flagships.
Barua et al. 2011. Defining Flagship Uses is Critical for Flagship Selection: A Critique of the IUCN Climate Change Flagship Fleet. <i>AMBIO: A Journal of the Human Environment</i> 40:431–435.	Flagship species	Selection criteria	Critique of proposed IUCN flagships and their lack of support or use. Recommendations are provided on how to improve selection and effectiveness.	New suite proposed by IUCN is not supported by potential effects and is haphazard.
Barua. 2011. Mobilizing metaphors: the popular use of keystone, flagship and umbrella species concepts. <i>Biodiversity and Conservation</i> 20:1427–1440.	Terminology	Use in articles	This paper draws from science communication studies and metaphor analysis, to examine how keystone, flagship and umbrella species concepts are used and represented in non-academic contexts. 557 news articles containing these terms were systematically analyzed. Number of articles explaining the terms keystone and flagship was low, and keystones were the most	(1) Communication is largely biased towards mammals, (2) everyday language plays a vital role in the interpretation of concepts, and (3) metaphors influence peoples’ actions and understanding. Conservation biologists need to engage with issues of language if public

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			misrepresented term. Keystones were metaphorically linked with balance, flagships with representation and umbrella species with protection.	conservation literacy is to be improved.
Branton & Richardson. 2011. Assessing the value of the umbrella-species concept for conservation planning with meta-analysis. Conservation Biology 25:9–20.	Umbrella species (but actually biodiversity indicators)	Criteria	Species richness and abundance of co-occurring species were consistently higher in sites where umbrella species were present than where they were not and for conservation schemes with avian than with mammalian umbrella species. There were no differences in species richness or species abundance with resource generalist or specialist umbrella species or based on taxonomic similarity of umbrella and co-occurring species. Taxonomic group abundance was higher in across-taxonomic umbrella species schemes than when umbrella species were of the same taxon as co-occurring species. Birds generally better umbrella species than mammals, with smaller birds generally better than bigger birds and omnivores better than herbivore/carnivore birds.	Almost all studies use avian or mammal umbrella species. Larger body size/home range does not translate to better umbrella species. Taxonomic (dis)similarity nor resource specialization nor trophic level translates to better umbrella species.
Isasi-Catala. 2011. Indicator, umbrellas, flagships and keystone species concepts: use and abuse in conservation ecology. Interciencia 36:31–38.	All	Use	There are different definitions for each of the categories of surrogate species, which has hindered their correct implementation. The main limitations are: i) confusion and ambiguity in definitions and classifications, ii) overstatement of their scope, iii) lack of a standard method for selecting species, iv) insufficient validation of the species and monitoring of the program, v) difficulties in implementation, and vi) insufficient biological information. Basically studies have shown	Surrogate species should be considered only as partial tools for assessing the degree of conservation of these systems. Combining the use of surrogate species with other assessment tools, could improve the effectiveness for perceiving and quantifying changes in biodiversity due to disturbances caused by

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			them to be 'effective' but not adequate; their use needs to be constrained and appropriate use needs to be better defined.	human activities.
Lindenmayer & Likens. 2011. Direct Measurement Versus Surrogate Indicator Species for Evaluating Environmental Change and Biodiversity Loss. <i>Ecosystems</i> 14:47–59.	All (primarily management and environmental indicators)	General use and validity	The steps in applying the indicator species approach are broadly similar to the direct measurement approach, except surrogacy relationships also must be quantified between a supposed indicator species or indicator group and the factors for which it is purported to be a proxy. Such quantification needs to occur via: (1) determining the taxonomic, spatial and temporal bounds for which a surrogacy relationship does and does not hold and (2) determining the ecological mechanisms underpinning a surrogacy relationship. The use of an indicator species approach needs to be better justified. Attempts to quantify surrogacy relationships may reveal that, in some circumstances, the alternative of direct measurement of particular entities of environmental or conservation interest will be the best option.	Vague justification of why particular entities are considered to be suitable indicator species, and also the conditions for which indicators are purported to be indicative, can undermine the validity of the concept. There is often a lack of transferability of a given indicator species or indicator group to other landscapes, ecosystems, environmental circumstances or sometimes over time in the same location.
Mellin et al. 2011. Effectiveness of biological surrogates for predicting patterns of marine biodiversity: a global meta-analysis. <i>Plos One</i> 6.	Biodiversity indicators and umbrella species	Effectiveness - Marine	Surrogate effectiveness was typically lower than generally assumed. The type of surrogate used was the strongest determinant of P, with higher taxa surrogates predicting higher P than all other types. Higher taxa approaches can provide valuable surrogates only at a scale where they reflect species-level patterns of beta diversity, and as long as the inherent uncertainty of taxonomic classifications tempers conclusions. Surrogates based on representation were less effective than those based on spatial	Surrogate effectiveness should be the greatest for higher taxa surrogates at a <10-km spatial scale, in low-complexity marine ecosystems such as soft bottoms, and using multivariate methods. Surrogate taxa should ideally have a broad distribution across different environments and incorporate many species with restricted distributions,

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Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			congruence of biodiversity metrics. Surrogate effectiveness varied among habitats and was lowest for tropical coral reefs. Probably the high bio- and functional diversity makes use of surrogates more difficult.	be feasible and cost-effective to identify and survey, and be amenable to survey at multiple spatial scales. Where the goal of a study is the prediction of biodiversity patterns, surrogacy methods based on spatial congruence should be used; where conservation planning is the goal, surrogacy methods based on representation are applicable
Murphy et al. 2011. A critical assessment of the use of surrogate species in conservation planning in the Sacramento-San Joaquin Delta, California (USA). <i>Conservation Biology</i> 25:873–878.	Biodiversity indicators and umbrella species	Validation, monitoring	The use of surrogate species, in the form of cross-taxon response-indicator species in the Sacramento-San Joaquin Delta, California was evaluated. There has been increasing reliance on surrogates in conservation planning for species listed under federal or state endangered species acts, although the agencies applying the surrogate species concept did not first validate that the surrogate and target species respond similarly to relevant environmental conditions.	Recently developed validation procedures may allow for the productive use of surrogates in conservation planning, but, used without validation, the surrogate species concept is not a reliable planning tool. Do not use them unless you validate their use first.
Sætersdal & Gjerde. 2011. Prioritising conservation areas using species surrogate measures: consistent with ecological theory? <i>Journal of Applied Ecology</i> 48:1236–1240.	Biodiversity indicators	Assumptions of use	Surrogate species approaches as biodiversity indicators were compared with with current knowledge on distribution patterns of species, as reflected in theories of community assembly. Assumptions necessary for successful functioning of surrogate species (nested species assemblages, cross-taxon congruence, spatio-temporal consistency) were evaluated with respect to predictions	The lack of a necessary scientific foundation may explain the disappointing results of empirical tests of surrogate species as biodiversity indicators. Site (reserve) selection should be based on costs and opportunities within



**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			from either niche or neutral community models. There is a general mismatch between ideas behind surrogate species and ecological community theory, except that surrogate species based on complementarity may be consistent with niche-based theory when gradients in species composition are strong.	complementary environmental/land units, rather than expensive inventories of unfounded surrogate species.
Veríssimo et al. 2011. Toward a systematic approach for identifying conservation flagships. <i>Conservation Letters</i> 4:1–8.	Flagship species	Definition, selection and verification	Flagship species are frequently used by conservation practitioners to raise funds and awareness for reducing biodiversity loss. 51 articles did not include a flagship definition and 30 articles used an incorrect definition of flagship species that mixed the characteristics of the concept with those of other surrogate species concepts. To reduce this problem, a new definition of “a species used as the focus of a broader conservation marketing campaign based on its possession of one or more traits that appeal to the target audience” is proposed that emphasizes their marketing role and includes an interdisciplinary framework to improve flagship identification, based on methodologies from social marketing, environmental economics, and conservation biology. The purpose of a campaign should be identified before working with the potential target audience to identify the most suitable species, and should monitor the success of their campaigns and feed this back into the marketing process. Return on investment analyses should be used to determine when funds are best spent on high-profile flagships and when raising the profile of other species	There are much needed improvements to current approaches for selecting flagship species so that they are underpinned by empirical evidence and conducted only after deciding the conservation target and identifying target stakeholders. A more effective evaluation of flagship species to increase the understanding of the concept’s strengths and weaknesses is needed. Such changes should ensure that the flagship approach is used more effectively to conserve a wider range of species and habitats.

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			is more appropriate. Flagship concept can be applied to other aspects of biodiversity, such as priority regions and species sharing specific traits.	
Heink & Kowarik. 2010. What are indicators? On the definition of indicators in ecology and environmental planning. <i>Ecological Indicators</i> 10:584–593.	Indicators	Definitions and terminology	Different meanings of the term "indicator" in ecology and environmental planning are evaluated. There are many ways an indicator is defined, but a broad definition is feasible. A general definition is suggested and recommendations for appropriate use are provided.	Indicators as ecological components and as measures should be distinguished from descriptive and normative indicators. To avoid problems based on different understandings of the term and to maintain integrity in its use, users should always provide a definition of the indicator term.
Heink & Kowarik. 2010. What criteria should be used to select biodiversity indicators? <i>Biodiversity and Conservation</i> 19:3769–3797.	Biodiversity indicators (actually more like surrogates generally although they refer to them all as biodiversity indicators)	Selection and verification of biodiversity indicators	This paper evaluates whether there are common approaches in selecting biodiversity indicators in ecology and environmental policy. The criteria used to select biodiversity indicators were evaluated to determine if they had been scientifically tested against their suitability. There are different patterns for selecting biodiversity indicators in the different fields of application. In ecology, the quality of indicators is mainly determined by a close relationship between indicator and indicated phenomenon. While the relevance of an indicator for a given issue or an assessment of a certain impact, is of paramount importance for conservation policy.	Few biodiversity indicators are empirically tested to determine if they meet the criteria by which they were purportedly chosen. To assess the suitability of a biodiversity indicator, it should be tested against all of the criteria relevant for its selection.
Heino. 2010. Are indicator groups and cross-taxon congruence useful for predicting biodiversity in aquatic	Biodiversity indicators	cross-taxon congruence in freshwater	Among popular surrogates are indicator groups that could be used for predicting variation in the biodiversity of other	As has been found in studies of terrestrial ecosystems, there is low utility for

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
ecosystems? Ecological Indicators 10:112–117.		systems	taxonomic groups. Despite some success at large scales, surveys of multiple taxonomic groups across ecosystems have suggested that no single group can be used effectively to predict variation in the biodiversity of other taxonomic groups. This paper evaluates indicator groups and cross-taxon congruence in species richness and assemblage composition patterns in inland aquatic ecosystems. Even when statistically highly significant correlations between taxonomic groups have been detected, these correlations have been too weak to provide reliable predictions of biodiversity among various taxonomic groups or biodiversity in general.	indicator groups in predicting the biodiversity of other taxa in aquatic ecosystems. Indicator groups and, more generally, cross-taxon congruence thus do not appear to be particularly relevant for conservation in the freshwater realm.
Lewandowski et al. 2010. The effectiveness of surrogate taxa for the representation of biodiversity. Conservation Biology 24:1367–1377.	Biodiversity indicators (species richness) for reserve selection?		A surrogate was 25% more likely to be effective with a complementarity approach than with a hotspot approach. For hotspot-based approaches, biome, extent of study, surrogate taxon, and target taxon significantly influenced effectiveness of the surrogate. For complementarity-based approaches, biome, extent, and surrogate taxon significantly influenced effectiveness of the surrogate. For all surrogate evaluations, biome explained the greatest amount of variation in surrogate effectiveness. Herpetofauna were the most effective taxon as both surrogate and target when a richness-hotspot approach was used; however, herpetofauna were include in few studies. For complementarity approaches, taxa that are feasible to measure and tend to have a large number of habitat specialists distributed	All of the published evaluations of surrogate taxa prior to 1999 used a richness hotspot approach. From 2000–2004, the proportion using a complementarity approach had grown to 42%. For 2005–2007 the proportion was 50%. This study focused only on large scales and single surrogate species. Surrogate taxa were most effective in grasslands and in some cases boreal zones, deserts, and tropical forests. Surrogate taxa also were more effective in studies examining larger areas.

**Table D-1. Summary of Review Papers**

Reference (Most Recent First)	Surrogate Reviewed	Portion of Process	Summary	Major Point(s)
			collectively across broad environmental gradients were the most effective surrogates.	
McArthur et al. 2010. On the use of abiotic surrogates to describe marine benthic biodiversity. <i>Estuarine Coastal and Shelf Science</i> 88:21–32.	Abiotic surrogates in marine systems	Use and measurement of abiotic surrogates for broad patterns of biodiversity	Abiotic surrogates of biodiversity are increasingly valuable in filling the gaps in knowledge of biodiversity patterns, especially identification of hotspots, habitats needed by endangered or commercially valuable species and systems or processes important to the sustained provision of ecosystem services. This review examines the use of abiotic variables as surrogates for patterns in benthic biodiversity with particular regard to how variables are tied to processes affecting species richness and how easily those variables can be measured at scales relevant to resource management decisions. Direct gradient variables can be strong predictive variables for larger systems, although local stability of water quality may prevent usefulness of these factors at fine spatial scales. Sediment variables often exhibit complex relationships with benthic biodiversity.	Pure spatial variables such as latitude, longitude and depth are not direct drivers of biodiversity patterns but often correspond with driving gradients and can be of some use in prediction. In such cases it would be better to identify what the spatial variable is acting as a proxy for so boundaries for that variable are not overlooked. The utility of these potential surrogates vary across spatial scales, quality of data, and management needs.

## APPENDIX D3: SUMMARY OF PRIMARY RESEARCH SINCE 2010

This reference list includes primary research on surrogate species from 2010 to early 2015.

Research prior to 2010 is generally not included, as it was assumed *Conservation by Proxy* (Caro 2010) and other reviews sufficiently summarize primary research prior to 2010. A tabular summary is also provided that summarizes the purpose, methods and conclusions of each paper.

- Bachand, M. et al. 2014. Species indicators of ecosystem recovery after reducing large herbivore density: comparing taxa and testing species combinations. *Ecological Indicators* **38**:12–19.
- Banks, J. E., A. S. Ackleh, and J. D. Stark. 2010. The use of surrogate species in risk assessment: using life history data to safeguard against false negatives. *Risk Analysis* **30**:175–182.
- Banks, J. E., J. D. Stark, R. I. Vargas, and A. S. Ackleh. 2014. Deconstructing the surrogate species concept: a life history approach to the protection of ecosystem services. *Ecological Applications* **24**:770–778.
- Barua, M., J. Tamuly, and R. A. Ahmed. 2010. Mutiny or Clear Sailing? Examining the Role of the Asian Elephant as a Flagship Species. *Human Dimensions of Wildlife* **15**:145–160.
- Beck, J., L. Pfiffner, L. Ballesteros-Mejia, T. Blick, and H. Luka. 2013. Revisiting the indicator problem: can three epigeal arthropod taxa inform about each other's biodiversity? *Diversity and Distributions* **19**:688–699.
- Bennett, J., R. Maloney, and H. Possingham. 2015. Biodiversity gains from efficient use of private sponsorship for flagship species conservation. *Proceedings of the Royal Society B-Biological Sciences* **282**:20142693.
- Breckheimer, I., N. Haddad, W. Morris, A. Trainor, W. Fields, R. Jobe, B. Hudgens, A. Moody, and J. Walters. 2014. Defining and evaluating the umbrella species concept for conserving and restoring landscape connectivity. *Conservation Biology* **28**:1584–1593.
- Butler, S., R. Freckleton, A. Renwick, and K. Norris. 2012. An objective, niche-based approach to indicator species selection. *Methods in Ecology and Evolution* **3**:317–326.
- Campos, F., J. Trindade-Filho, D. Brito, G. Llorente, and Solé. 2014. The efficiency of indicator groups for the conservation of amphibians in the Brazilian Atlantic Forest. *Ecology and Evolution* **4**:2505–2514.
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- Castellón, T. D., and K. E. Sieving. 2012. Can Focal Species Planning for Landscape Connectivity Meet the Needs of South American Temperate Rainforest Endemics? *Natural Areas Journal* **32**:316–324.
- Che-Castaldo, J. P., and M. C. Neel. 2012. Testing Surrogacy Assumptions: Can Threatened and Endangered Plants Be Grouped by Biological Similarity and Abundances? *Plos One* **7**:e51659–e51659.
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- Cushman, S., and E. Landguth. 2012. Multi-taxa population connectivity in the Northern Rocky Mountains. *Ecological Modelling* **231**:101–112.
- Di Minin, E., and A. Moilanen. 2014. Improving the surrogacy effectiveness of charismatic megafauna with well-surveyed taxonomic groups and habitat types. *Journal of Applied Ecology* **51**:281–288.
- Eglington, S. M., D. G. Noble, and R. J. Fuller. 2012. A meta-analysis of spatial relationships in species richness across taxa: birds as indicators of wider biodiversity in temperate regions. *Journal for Nature Conservation* **20**:301–309.
- Epps, C. W., B. M. Mutayoba, L. Gwin, and J. S. Brashares. 2011. An empirical evaluation of the African elephant as a focal species for connectivity planning in East Africa. *Diversity and Distributions* **17**:603–612.
- Esselman, P., M. Edgar, J. Breck, E. Hay-Chmielewski, and L. Wang. 2013. Riverine connectivity, upstream influences, and multi-taxa representation in a conservation area network for the fishes of Michigan, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems* **23**:7–22.
- Fattorini, S., R. L. H. Dennis, and L. M. Cook. 2011. Conserving organisms over large regions requires multi-taxa indicators: one taxon's diversity-vacant area is another taxon's diversity zone. *Biological Conservation* **144**:1690–1701.
- Grantham, H., R. Pressey, J. Wells, and A. Beattie. 2010. Effectiveness of Biodiversity Surrogates for Conservation Planning: Different Measures of Effectiveness Generate a Kaleidoscope of Variation. *PLoS ONE* **5**:e11430.
- Hermoso, V., S. R. Januchowski-Hartley, and R. L. Pressey. 2013. When the suit does not fit biodiversity: loose surrogates compromise the achievement of conservation goals. *Biological Conservation* **159**:197–205.
- Hoare, J. M., A. Monks, and C. F. J. O'Donnell. 2012. Can correlated population trends among forest bird species be predicted by similarity in traits? *Wildlife Research* **39**:469–477.
- Hoare, J. M., A. Monks, and C. F. J. O'Donnell. 2013. Do population indicators work? Investigating correlated responses of bird populations in relation to predator management. *Ecological Indicators* **25**:23–34.

- James, C. A., J. Kershner, J. Samhouri, S. O'Neill, and P. Levin. 2012. A Methodology for Evaluating and Ranking Water Quantity Indicators in Support of Ecosystem-Based Management. *Environmental Management* **49**:703–719.
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- Neeson, T. M., I. Van Rijn, and Y. Mandelik. 2013. How taxonomic diversity, community structure, and sample size determine the reliability of higher taxon surrogates. *Ecological Applications* **23**:1216–1225.
- Nekaris, K., A. Arnell, and M. Svensson. 2015. Selecting a Conservation Surrogate Species for Small Fragmented Habitats Using Ecological Niche Modelling. *Animals* **5**:27–40.
- Nicholson, E., D. B. Lindenmayer, K. Frank, and H. P. Possingham. 2013. Testing the focal species approach to making conservation decisions for species persistence. *Diversity and Distributions* **19**:530–540.
- Noon, B. R., L. L. Bailey, T. D. Sisk, and K. S. McKelvey. 2012. Efficient species-level monitoring at the landscape scale. *Conservation Biology* **26**:432–441.

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- Shanley, C., S. Pyare, and W. Smith. 2013. Response of an ecological indicator to landscape composition and structure: Implications for functional units of temperate rainforest ecosystems. *Ecological Indicators* **24**:68–74.
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**Table D-2. Summary of Primary Research (since 2010)**

Reference (Most Recent First)	>Stated Surrogate< Purpose	Methods	Conclusion(s)
<p>Bennett et al. 2015. Biodiversity gains from efficient use of private sponsorship for flagship species conservation.</p>	<p>&gt;Flagship species&lt; Using a case study of flagship species sponsorship programs in New Zealand, we quantify the potential biodiversity gains from private sponsorship of flagship species and determine whether further gains can be achieved when sponsorship of flagship species is integrated with a cost-effectiveness approach. Specifically, we incorporate funding for flagship species conservation into a prioritization protocol for NZ's 700 most threatened species, to answer the following questions: 1) what are the biodiversity gains from privately-funded flagship species conservation, compared to using the same funds in a strictly cost-effectiveness approach; and 2) what are the biodiversity gains when flagship funding is combined with a cost-effectiveness approach to selectively fund flagship species conservation actions that maximally benefit other threatened species?</p>	<p>Used an existing protocol which ranks potential recovery projects for 700 of the most threatened species in New Zealand based on their cost effectiveness. Ten flagship species could receive special funding and for each funding scenario, we ran the prioritization protocol using a range of baseline budgets from \$5M to \$50M NZD per year. We used two measures to quantify biodiversity gains: the number of additional threatened species that could get funding and the estimated additional phylogenetic diversity that could be gained by conserving these threatened species.</p>	<p>Randomly allocating funding among actions for each of the ten flagship species allowed gains of up to five additional threatened species over the baseline scenario of no additional investment. Allocating this funding to actions that maximized benefits to other species resulted in gains of up to six species beyond the baseline scenario. Allocating this funding to efficient actions from a larger suite of 22 flagship species also resulted in gains of up to six species, and generally greater biodiversity gains. The scenario where the extra funding was donated for general biodiversity goals and applied directly to the prioritization protocol rather than to flagship species resulted in the greatest biodiversity gains with benefits up to 13 additional species compared to the baseline scenario, and a marked increase in phylogenetic diversity. Private funding for flagship species can clearly result in additional species and phylogenetic diversity conserved, via conservation actions shared with other species. Gains from funding only flagship species are consistently smaller than scenarios where private funding could be optimally allocated among all threatened species.</p>

**Table D-2. Summary of Primary Research (since 2010)**

Reference (Most Recent First)	>Stated Surrogate< Purpose	Methods	Conclusion(s)
<p>McDermid et al. 2015. Identifying a suite of surrogate freshwaterscape fish species: a case study of conservation prioritization in Ontario’s Far North, Canada.</p>	<p>&gt;Landscape species&lt; Given the importance of freshwater resources in Ontario Far North (OFN), existing information on fish species was compiled and assessed using the Landscape Species Approach. This approach is a species-based conservation planning tool developed for terrestrial conservation that is constructed around the identification of focal species for a given landscape. This paper presents the first application of this approach to freshwater systems.</p>	<p>Using the information from five criteria, lake sturgeon, lake trout, and walleye were selected as landscape species. An analysis of 14 large-bodied candidate freshwater species, including their area requirements, habitat use, ecological function, socio-economic function and vulnerability to threats was used to create a landscape design for these three species.</p>	<p>The LSA method identified mining, road access, and climate change as the three most severe threats to the freshwaterscape. The identification of these species and their ecological requirements suggests a starting place for research, management, and conservation of freshwater resources in OFN before large-scale landscape changes. Unfortunately, data was available for only 14 large-bodied candidate species and habitats in lake environments, so there remains a need for information on small-bodied species and other lower trophic level organisms. There was also an almost complete lack of information on distributions of river dwelling species.</p>
<p>Nekaris et al. 2015. Selecting a Conservation Surrogate Species for Small Fragmented Habitats Using Ecological Niche Modelling.</p>	<p>&gt;Flagship and umbrella&lt; We aim to find a suitable species among the less charismatic animal species left in the fragmented forests of South-western Sri Lanka. We selected ten candidates, using a questionnaire survey along with computer modelling of their distributions. The red slender loris and the fishing cat came out as finalists as they were both appealing to local people, and fulfilled selected ecological criteria.</p>	<p>We applied a set of selection criteria to all mammals with relatively forward facing large eyes within the study area to produce a shortlist of 10 species. Two species were selected based on a survey of local perceptions. We tested for umbrella characteristics in the original shortlist, utilizing Maximum Entropy (MaxEnt) modelling, and analysed distribution overlap. We recruited participants who would support wildlife, but at a local level, from target groups of Sri</p>	<p>Combining local opinion with ecological niche modelling on mammals in highlighted the red slender loris as the most appropriate surrogate species candidate. From the survey, both finalists were well-regarded. Both candidates exhibited the typically favoured characteristic of forward-facing eyes. Red slender loris, however, achieved higher scores for all three umbrella criteria and was selected as a more ideal surrogate species for this highly fragmented forest network.</p>

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		Lankan residents.	
Bachand et al. 2014. Species indicators of ecosystem recovery after reducing large herbivore density: Comparing taxa and testing species combinations.	>Indicator species (management)< The objectives of this study are (1) to assess the complementary value of plants, insects and songbirds as potential indicator species for monitoring ecosystem recovery after reducing deer densities and (2) to verify, using plants as a model taxon, whether species combinations can be more efficient indicators of ecosystem recovery than single species.	Compared abundance, percent cover, indicator value, as well as single versus multiple species surrogates. Wildlife treatments (full factorial split plot) were implemented and then collected data on species in each plot.	Moths (for uncut) and plants (for cutover) are best indicators. Birds, bees and carabid beetles were not good indicators. Species combinations were better at representing more ecosystem states.
Banks et al. 2014. Deconstructing the surrogate species concept: a life history approach to the protection of ecosystem services.	>Surrogate species relative to toxicity (environmental indicator)< Evaluate effects of environmental toxins on arthropods other than honeybees. Verify if surrogacy assumptions hold.	Modelled population dynamics of 4 species of parasitoid wasps (that provide ecosystem services). Modelling survivorship and fecundity reductions. Modeling only	3 of the species represented each other, while the 4th species functioned very differently. Based on the modeling here, using one of the 3 species as an indicator would work somewhat.
Breckheimer et al. 2014. Defining and Evaluating the Umbrella Species Concept for Conserving and Restoring Landscape Connectivity.	>Umbrella species (connectivity)< Identify a potential umbrella species for planning habitat connectivity in a landscape with already defined conservation areas. We define a good connectivity umbrella as a species for which conservation or restoration of its dispersal habitat also facilitates dispersal of other target species.	Used actual habitat use data combined with model for dispersal to develop potential dispersal habitat. Compared this potential dispersal habitat among 3 species to determine which one served as the best umbrella, with the assumption the bird would be the best since it has the largest home ranges. We tested this assumption by developing a quantitative method to measure overlap in dispersal habitat of 3 threatened species, bird (the umbrella), a	Despite differences in natural history and breeding habitat, we found substantial overlap in the spatial distributions of areas important for dispersal of this suite of taxa. However, the intuitive umbrella species (the bird) did not have the highest overlap with other species in terms of the areas that supported connectivity. The red-cockaded woodpecker was NOT a good umbrella species, although it is already used for this purpose in this landscape. Gopher frog was the best umbrella species of

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		butterfly, and a frog, inhabiting the same fragmented landscape. Actually measured movements of animals.	the three.
Campos et al. 2014. The efficiency of indicator groups for the conservation of amphibians in the Brazilian Atlantic Forest.	>Biodiversity indicator< Using a constructive approach, the main purposes of this study were to evaluate the performance and efficiency of eight potential indicator groups representing amphibian diversity in the Brazilian Atlantic Forest.	Data on the geographic range of amphibian species that occur in the Brazilian Atlantic Forest were overlapped to the full geographic extent of the biome, which was divided into a regular equal-area grid. Optimization routines based on the concept of complementarity were applied to verify the performance of each indicator group selected in relation to the representativeness of the amphibians in the Brazilian Atlantic Forest as a whole, which were solved by the algorithm “simulated annealing,” through the use of the software MARXAN.	Some indicator groups were substantially more effective than others in regard to the representation of the taxonomic groups assessed. Leiuperidae was considered the best indicator group among the families analyzed, representing 71% of amphibian species in the Brazilian Atlantic Forest, which may be associated with the diffuse geographic distribution of their species. In this sense, this study promotes understanding of how the diversity standards of amphibians can be informative for systematic conservation planning on a regional scale.
Di Minin & Moilanen. 2014. Improving the surrogacy effectiveness of charismatic megafauna with well-surveyed taxonomic groups and habitat types.	>Umbrella species < Analyzed potential umbrella species. The analysis included the ‘Big Five’ (lion, leopard, elephant, buffalo, black rhino and white rhino), as well as other mammals and 288 invertebrate species belonging to 10 different families.	A total of 662 biodiversity features, including habitat types, and species and populations from six taxonomic groups, were used. The habitat types included in this study have earlier also been used as surrogates for specific ecological processes. We started by developing a spatial conservation prioritization for the ‘Big Five’ only and then investigated how supplementing	Other taxa are not good surrogates for charismatic mammal species. Habitat types are a necessary component of surrogacy strategies that cover plants and insects. Overall, a combination of habitat types and charismatic mammals, complemented with other well-known taxa (birds, amphibians and reptiles), provided the highest surrogacy effects. Generally, our results confirm the low capacity of one taxonomic group to predict priority

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		<p>data on other well-surveyed taxa helped improving their surrogacy effectiveness. Meanwhile, we developed a spatial conservation prioritization for habitat types only and then investigated how supplementing data on well-surveyed taxa, including the ‘Big Five’, helped improving the surrogacy effectiveness of habitat types.</p>	<p>areas for other targeted taxa.</p>
<p>Kanagavel et al. 2014. Beyond the “General Public”: Implications of Audience Characteristics for Promoting Species Conservation in the Western Ghats Hotspot, India.</p>	<p>&gt;Flagship species&lt; In this study, we seek to determine from a multi-stakeholder perspective, audience characteristics that influence perceptions towards wildlife at Valparai, a fragmented plateau in the Western Ghats region of the Western Ghats-Sri Lanka Hotspot.</p>	<p>We started by selecting 18 species known to occur in the Anamalai Hills as to represent a wide range of taxonomic groups, physical appearances, IUCN threat status and local cultural values. A questionnaire to assess respondents’ attitudes towards different species was then developed which presented each respondent with color photographs of a randomly selected subset of six of the above-mentioned species. Respondents were then asked to rate each species on a five-point likertscale (“strongly like”, “like”, “neutral”, “dislike”, or “strongly dislike”).</p>	<p>Overall, the Indian peafowl, Great Hornbill, and lion-tailed macaque were the most-liked species. Overall, the tiger and elephant, two of the most widely used conservation flagship species, were placed lower in the preference ranking probably as a result of human-wildlife conflicts. The relatively high score received by the elephant even though it was involved in human-wildlife conflicts suggests that the cultural and religious ties associated with the species allows for continued positive appreciation. As expected, species that are often considered less aesthetically attractive, perceived as bad omens and/or as a threat feature lower in the ranking. On the other hand, the high overall ranking of the southern birdwing, the largest south Indian butterfly, Parachuting frog and the</p>

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			Deccan mahseer, an endangered freshwater fish reinforce the notion that usually neglected taxonomic groups can be used as conservation flagships. We found that stakeholder group membership was the most important characteristic followed by gender. Our results emphasize the need to design conservation campaigns with specific audiences in mind, instead of the very often referred to “general public”.
Lindenmayer et al. 2014. An Empirical Assessment and Comparison of Species Based and Habitat-Based Surrogates: A Case Study of Forest Vertebrates and Large Old Trees.	>Indicator species< This study examined (1) the effectiveness of the abundance of a particular species of arboreal marsupial as a species-based surrogate for other species of arboreal marsupials, (2) the effectiveness of the abundance of hollow-bearing trees as a habitat surrogate for the abundance of particular species of arboreal marsupials, (3) if a particular class of surrogate was consistently better than the other broad class of surrogate over the 30- year time frame of our work, and (4) whether a combination of both species and habitat surrogates performed better than either kind of proxy in isolation.	Used abundance of surrogate species (1 species) and counted hollow trees. Compared various modeled outputs and data from habitat use.	We found a significant positive association of the abundance of both target species with the habitat-based surrogate in all four datasets, irrespective of whether the species-based surrogate was (or was not) included in the model. However, the association with the species-based surrogate was significant in only one case. The habitat-based surrogate was therefore 10 times less effort (and hence substantially less costly) to measure than the species-based surrogate.
Myšák & Horsák 2014.	>Surrogate species - for reserve		We found that spatial congruence

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Biodiversity surrogate effectiveness in two habitat types of contrasting gradient complexity.	selection< We assessed both species richness and community composition to evaluate cross-taxon congruence between vascular plants, nonvascular plants and lichens, and terrestrial snails. We tried to answer these questions: (1) What is the effect of habitat type and gradient complexity on community structure patterns? (2) Can vascular plants serve as universal surrogate taxon for nonvascular plants and lichens, and snails in conservation planning regardless to habitat type? (3) Is there congruence in species richness and number of at-risk species?		among studied taxa was affected by habitat type, however vascular plants were good indicator of snail biodiversity in both habitats. Nevertheless, all significant positive correlations of species richness were associated with the congruence in main environmental gradients. Although there was a consistency in significantly positive cross-taxon correlation in community similarity, the congruence was insufficient for conservation purposes. Furthermore we confirmed the necessity of integration of at-risk species in conservation planning as Red List species were poor indicators for total species richness and vice versa. We suggest the complementation of existing reserve network with small scale protected areas focused on conservation of at-risk ecosystems, communities or species. In this study vascular plants were not found as a sufficient indicator for fine-filter conservation of other taxa.
Olds et al. 2014. Incorporating Surrogate Species and Seascape Connectivity to Improve Marine Conservation Outcomes.	>Flagship, umbrella species, and landscape species< The bumphead parrotfish was used as a multi-faceted surrogate species (flagship, umbrella and landscape) to create a marine reserve design in the Solomon	To test the effectiveness of the conservation strategy implemented, we explicitly accounted for seascape connectivity in the study design by testing the secondary hypothesis that seascape	The local utility of bumphead parrotfish as a surrogate species likely relates to their habitat use through ontogeny rather than the requirements of adults. Adults have large home ranges on offshore reefs and, therefore, are potentially best

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	Islands. The effectiveness of this marine reserve design was tested for benefits to other species.	connectivity among seagrasses, mangroves, and coral reefs would enhance the ability of reserves to promote fish abundance (including bumphead parrotfish, which use a range of different habitats through ontogeny). We first determined that bumphead parrotfish were more abundant in reserves than adjacent fished waters. We then assessed whether bumphead parrotfish were an effective surrogate for multispecies conservation by testing our primary hypothesis that fish assemblages would follow patterns in bumphead parrotfish abundance and, therefore, differ between the reserves and adjacent fished locations.	managed with alternative strategies to marine reserves (e.g., spearfishing bans and protection of aggregation sites). Seascape connectivity correlates with improved performance of marine reserves and supports the case for greater incorporation of spatial ecology into management of reefs and adjacent areas as functional seascapes. Due to the spatially heterogeneous influence of seascape connectivity on reserve performance, it can really matter how and where reserves are monitored to assess their effectiveness. For example, we examined reefs that were both close to and isolated from adjacent seagrass and mangroves and detected only strong reserve effects on reefs near adjacent habitat. If only isolated reefs were examined, the conclusion may have been different.
Rickbeil et al. 2014. Assessing conservation regionalization schemes: employing a beta diversity metric to test the environmental surrogacy approach.	>Surrogate species - for reserve selection< This study examined three questions: (1) do environmental regionalizations effectively delineate avian communities, and how does their performance compare to a regionalization built using species data directly? (2) How is the performance of regionalizations affected by	We employed a beta diversity metric using community data from the British Columbia Breeding Bird to assess the ability of a number of environmental regionalization schemes to delineate species turnover. We also developed a new species-based scheme using kriged local beta diversity values and a thematic resolution optimized	All regionalization schemes delineated significant patterns in community structure, with the Bird Conservation Regions performing most similarly to the species-based regionalization. Our results indicate that avian communities are structured both environmentally and spatially, with all regionalization schemes delineating significant patterns in beta diversity.



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	enforcing spatial contiguity within regions? (3) What effect does changing thematic resolution have on regionalization performance?	through ANOSIM testing, which was then evaluated against the previously tested schemes. Each conservation regionalization scheme was tested in terms of its ability to delineate patterns of beta diversity exhibited by the Bird Atlas data.	Environmental regionalizations can function as effective alternatives to species-based regionalizations, particularly in areas with poor availability of species data. Also, we conclude that spatially contiguous regionalizations are superior to non-contiguous ones for delineating distinct communities. Lastly, we demonstrate how thematic resolution represents a trade-off between overall regionalization performance and regional redundancy, and how differing thematic resolutions can be employed depending upon the goals of the user. Common spatial and environmental patterns were observed between avian and butterfly communities.
Veríssimo et al. 2014. Using a Systematic Approach to Select Flagship Species for Bird Conservation.	>Flagship species< We used a systematic and stakeholder-driven approach to select flagship species for a conservation campaign in the Serra do Urubu in northeastern Brazil. We used an audience-driven flagship-selection approach, underpinned by marketing theory, to identify a new flagship bird for the Serra do Urubu. This flagship species will be used in future marketing campaigns to raise conservation awareness. We also	We based our techniques on environmental economic and marketing methods. We used choice experiments to examine the species attributes that drive preference and latent-class models to segment respondents into groups by preferences and socioeconomic characteristics. We used respondent preferences and information on bird species inhabiting the Serra do Urubu to calculate a flagship species suitability score.	The species' traits that drove audience preference were geographic distribution, population size, visibility, attractiveness, and survival in captivity. However, the importance of these factors differed among groups and groups differed in their views on whether species with small populations and the ability to survive in captivity should be prioritized. The popularity rankings of species differed between approaches, a result that was probably related to the different ways in which the 2 methods measured

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	compared this approach with a plurality flagship vote, a commonly used approach to flagship selection.		preference. Our new approach is a transparent and evidence-based method that can be used to refine the way stakeholders are engaged in the design of conservation marketing campaigns.
Veríssimo et al. 2014. Evaluating Conservation Flagships and Flagship Fleets.	>Flagship species< We develop an evaluation strategy for conservation flagships, and use it to: measure the effectiveness of an existing bird flagship species; detect whether additional species are needed; and, if appropriate, identify which species should be added to create a flagship fleet. We apply these methods to a conservation project in Brazil to: (1) measure the impact of an ongoing conservation outreach project to raise the profile of a flagship species and (2) determine whether this flagship matches the preferences of the target audience and if it should be supplemented with other species to form a flagship fleet.	To assess the Araripe manakin’s role as a conservation flagship, we first needed to identify the target audience. We selected the rural communities living adjacent to the species habitat, as the main threat to the species is habitat degradation due to subsistence-level resource use by local villagers. Used Latent Class Model to group the stakeholders.	We found the Araripe manakin is currently an effective conservation flagship in terms of target audience visibility and recognition, but has traits that appeal to only half the target audience. We also show that this shortcoming could be overcome by forming a flagship fleet based on adding an endemic mammal or fish species but there are additional strategic considerations that must be taken into account, namely in terms of costs and potential future conflicts. A high proportion of respondents’ recognized their own role in the conservation of the Araripe manakin by naming local communities as key players in the species conservation. Only two of the three target audience groups said that endemic bird species were appealing. The remaining group favored endemic mammals as a flagship, together with a weaker preference for fish species.
Beck et al. 2013. Revisiting the indicator problem: can three epigeal arthropod taxa inform	>Biodiversity indicator< Using three arthropod taxa that share the same habitat, utilize	We used a large data set of pitfall trap samples from different habitats for three taxonomic	We found positive, yet not very strong, correlations in biodiversity patterns, while environmental models differed

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about each other's biodiversity?	similar resources and are sampled with identical technique, we investigate the applicability of two levels of biodiversity indication: (1) prediction of biodiversity patterns, and (2) inference of environment–biodiversity relationships. The second aspect is of high relevance to applied conservation management yet mostly neglected, at least in terrestrial systems, when discussing the indicator concept.	groups (Carabidae, Staphylinidae and Araneae). We quantified biodiversity by different metrics of local diversity (species richness, effective number of species) and of pairwise faunal dissimilarities (Sørensen, Bray–Curtis). We investigated the congruence of (1) biodiversity patterns by cross-taxa regressions, and (2) environmental models of biodiversity by comparing fitted coefficients, and resulting extrapolations across the research region.	considerably between taxa as well as between diversity metrics. Inferences of environment–biodiversity relationships can differ between taxonomic groups even if biodiversity patterns alone show significant correlation. This may be either because species indeed respond differently to environmental variation or because of misspecifications inherent in ecological modelling. Both possibilities suggest a need for caution in selecting and applying biodiversity indicators. Furthermore, the choice of diversity metric can strongly affect results, and therefore, decisions about which metric to use in any given situation need to be made carefully.
Esselman et al. 2013. Riverine connectivity, upstream influences, and multi-taxa representation in a conservation area network for the fishes of Michigan, USA.	>Umbrella species (Marxan)< Landscape spatial planning to identify focal areas for conservation of fishes is an important step to targeted site-level interventions to protect or restore fish habitats, because it can provide a strategic approach to guide conservation efforts.	A commonly used systematic planning software, Marxan, was employed with previously published fish range and human disturbance predictions to define a network of conservation focal areas for rivers in Michigan. This network focused on large-bodied species, small-bodied species, species of greatest conservation need (SGCN), and all species together.	Depending on the scenario, the networks identified comprised between 14 and 20% of Michigan stream length in over 1700 focal areas. Mean focal area sizes were much larger for the Upper Peninsula than the Lower Peninsula. Approximately 35% of the focal areas defined flowed through already protected lands, but less than 5% had upstream catchments that were secure within protected areas. There was a 45% overlap in the focal areas selected for the large- and small-bodied fish and SGCN. Resultant

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			maps show locations with a high natural potential to conserve all of Michigan’s native fish species, and can serve as a reference point for comprehensive state-wide planning efforts for fish conservation.
<p>Hermoso et a. 2013. When the suit does not fit biodiversity: Loose surrogates compromise the achievement of conservation goals.</p>	<p>&gt;Coarse-filter surrogate&lt; We evaluate the role of three factors that could affect the effectiveness of coarse-filter surrogates: (a) thematic resolution (number of classes), (b) species’ prevalence, and (c) the ability of classifications to portray homogeneous communities (classification strength). We explore the role of direct and indirect effects of these factors with a simulated dataset of 10,000 planning units and 96 species and structural equation modelling.</p>	<p>We use a simulated dataset to compare the performance of different coarse-filter surrogates in a system with known properties. We test three levels of thematic resolution and three different levels of prevalence (common, intermediate and rare). Additionally our simulated dataset contains a wide range of classification strength conditions, from strong classifications with no between-class species overlap and high within-class homogeneity to weak classifications with classes sharing most species.</p>	<p>Three different factors (classification strength, thematic resolution and species prevalence) can independently and interactively influence the effectiveness of coarse-filter surrogates. The effectiveness of coarse-filter surrogates improved by increasing thematic resolution and with higher numbers of classes. With the finest thematic resolution and consistently high species prevalence, effectiveness was only better than random when classification strength &gt;0.5. Our results show that two main strategies could be followed to improve the value of classifications as coarse-filter surrogates: (i) refine thematic resolution, but more importantly (ii) improve the capacity to portray biodiversity patterns. A strategy under these circumstances would be to incorporate new variables that better explain current biodiversity patterns in conjunction with better knowledge of the environmental drivers of biodiversity.</p>

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<p>Hoare et al. 2013. Do population indicators work? Investigating correlated responses of bird populations in relation to predator management.</p>	<p>&gt;Management indicator&lt; Monitor forest birds to assess exotic mammal management. Specifically, we investigate whether (1) life history traits can be used to predict and negative correlations between trends in forest bird species, and (2) control of introduced mammals influences positive correlations between bird populations.</p>	<p>We evaluated population trends in 21 bird species vulnerable to predation by introduced mammals (primarily mustelids and rodents) at managed and unmanaged beech forest sites. Point counts, breeding traits, single site models, Bayesian meta analysis were used.</p>	<p>We found little support for the notion that life history traits can predict population indicator capabilities, either at sites where introduced mammalian predators are managed or at control sites. Neither positive nor negative correlations in species trends could be predicted based on life history traits and predator management did not produce consistent, correlated population trends among sites. Our results do not support the use of a population indicator approach to management for forest birds in New Zealand.</p>
<p>Neeson et al. 2013. How taxonomic diversity, community structure, and sample size determine the reliability of higher taxon surrogates.</p>	<p>&gt;Biodiversity indicator and umbrella species (reserve selection)&lt; The purpose of the study is to test the utility of 'higher taxon' surrogates and develop some guidelines for their use as biodiversity indicators (and somewhat for reserve selection). Specifically, we examined three aspects of the efficiency of the higher taxa approach: (1) the correlations between species richness, genus richness, and family richness, including potential sources of spurious correlation; (2) the relative shapes and asymptotes of species, genus, and family</p>	<p>We developed a mathematical model to show how taxonomic diversity, community structure, and sampling effort together affect three measures of higher taxon performance: the correlation between species and higher taxon richness, the relative shapes and asymptotes of species and higher taxon accumulation curves, and the efficiency of higher taxa in a complementarity-based reserve-selection algorithm. We performed a series of computer simulation experiments using this model to describe how taxonomic diversity, community structure, and sampling effort together influence</p>	<p>Higher taxon surrogates performed well in communities in which a few common species were most abundant, and less well in communities with many equally abundant species. Furthermore, higher taxon surrogates performed well when there was a small mean and variance in the number of species per higher taxa. We also show that empirically measured species-higher-taxon correlations can be partly spurious (i.e., a mathematical artifact), except when the species accumulation curve has reached an asymptote. We found that higher taxon surrogates were an efficient basis for selecting reserve networks, relative to cross-taxon</p>

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	accumulation curves; and (3) the performance of higher taxa in a simple site selection algorithm for the design of a reserve network.	the performance of higher taxon methods.	surrogates and environmental surrogates. Higher taxon surrogates in our model always outperformed environmental surrogates, suggesting that higher taxon surrogates may be a more useful type of surrogate. Genus level surrogates in our model usually outperformed for cross-taxon surrogates, but family-level surrogates were usually less efficient than cross-taxon surrogates.
Nicholson et al. 2013. Testing the focal species approach to making conservation decisions for species persistence.	>Focal species (Lambeck)< Reserve selection based on minimizing the expected loss of species, by estimating the risk of extinction with a metapopulation model - using 3 species as focal species representing 10 species.	First, found reserve for each species separately. Second, found reserve that minimized extinctions across all species. To test the subset of focal species, found the reserve that minimized extinctions for 3 focal species. And found best reserve for all 120 sets of three species, and compared them with 10-species reserve solution to test if any other three-species combination would deliver the same result.	It is valuable to include more than 1 focal species. The reserve solution that minimized the expected number of extinctions across 3 focal species was same as reserve solution that minimized the expected extinctions across all 10 species. The best multiple-species reserve solution differed from any of the single-species solutions. The extinction risk of each focal species was higher in the multiple-species reserve solutions than in the optimal single-species reserve solutions. The multispecies solution was a compromise reserve system that balanced the needs of >1 species.
Schindler et al. 2013. Multiscale performance of landscape metrics as indicators of species of plants, insects and vertebrates richness.	>Biodiversity indicator< Using Mediterranean forest landscape, Dadia National Park (Greece), as a case study area, we explored the performance of 52	We computed the landscape metrics for circular areas of five different extents around each of 30 sampling plots. We applied linear mixed models to evaluate	Our results showed that landscape metrics were good indicators for overall species richness, woody plants, orthopterans and reptiles. Metrics quantifying patch shape, proximity,

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	<p>landscape level landscape metrics as indicators of species richness for six taxa (woody plants, orchids, orthopterans, amphibians, reptiles, and small terrestrial birds) and for overall species richness.</p>	<p>significant relations between metrics and species richness and to assess the effects of the extent of the considered landscape on the performance of the metrics. Compared multiple models for using landscape metrics to estimate species richness.</p>	<p>texture and landscape diversity resulted often in well-fitted models, while those describing patch area, similarity and edge contrast rarely contributed to significant models. Spatial scale affected the performance of the metrics, since woody plants, orthopterans and small terrestrial birds were usually better predicted at smaller extents of surrounding landscape, and reptiles frequently at larger ones. Virtually no significant relations were detected between the metrics and species richness of orchids or amphibians.</p>
<p>Shanley et al. 2013. Response of an ecological indicator to landscape composition and structure: Implications for functional units of temperate rainforest ecosystems.</p>	<p>&gt;Ecological indicator&lt; We evaluated the northern flying squirrel, a known associate of key habitat features and processes of old-growth forest, for its capacity as a broad-scale ecological indicator of temperate rainforest ecosystem condition in southeastern Alaska, USA.</p>	<p>We utilized a spatially explicit, resource-selection function to evaluate its distribution relative to landscape composition and structure at local (within home-range) and broad (home-range selection) spatial scales, followed by a moving-window analysis to model patch occupancy across this landscape.</p>	<p>We found strong support for the influence of type, size, and compositional elements: large, old-growth patches were selected at both spatial scales, and regenerating forest patches <math>\leq 40</math> yrs old were selected against at the broader scale. More importantly, we found that occupancy was related to critical thresholds in composition: patches required <math>\geq 73\%</math> old-growth forest cover or a minimum total area of 73 ha of old-growth forest to be occupied by flying squirrels. These results are consistent with recent studies of this and related species and suggest that occurrence of northern flying squirrels in southeastern Alaska is influenced by</p>

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			a number of landscape structure and compositional variables that relate critically to late-seral forest conditions.
Tulloch et al. 2013. Accounting for Complementarity to Maximize Monitoring Power for Species Management.	>Management indicator < Our objective was to design an optimal monitoring program for determining the effectiveness of a management action. We sought the best set of indicator species to monitor that would maximize the likelihood of detecting a meaningful change in the target species for a given budget. Species to be monitored (i.e., indicator species) and species that are in need of conservation (i.e., target species) can be the same. We used 2 groups of target species: all mammals in the system that we believe are affected by the management action and only mammals listed as threatened.	Decision-science approach to cost-effective monitoring consisted of 6 steps: define monitoring objectives and constraints; list candidate indicators and calculate costs of monitoring each; define data underlying species responses to management and determine likelihood of detecting a trend; determine surrogacy value; combine information on trend detection and surrogacy to calculate monitoring benefits; and solve optimization problems. Did not assess with field data going forward, only retrospective analysis of long-term data set.	Basically using this approach resulted in a slightly different selection of indicator species than currently used, which provides more information at less cost for monitoring the effectiveness of fox control.
Butler et al. 2012. An objective, niche-based approach to indicator species selection.	>Indicator species< We present an objective, niche-based approach for species' selection, founded on a coarse categorisation of species' niche space and key resource requirements, which ensures the resultant indicator has these key attributes.	We use UK farmland birds as a case study to demonstrate this approach, identifying an optimal indicator set containing 12 species. We used outputs from risk assessments to test how representative the response to land-use change of each indicator set is of the response of the wider community to the same changes.	When applied to UK farmland birds, we identified an optimal indicator set containing 12 species. We show that the niche space occupied by these species fully encompasses that occupied by the wider community of 62 species and that their response to change is a strong correlate to that of the wider community. Furthermore, the temporal dynamics of an index



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		Secondly, we used national population trend data between 1970 and 2006 to calculate annual values for FBIall and an index based on the species set identified earlier and compared their temporal dynamics. Thirdly, we used data from the Breeding Bird Survey (BBS) (Risely, Noble & Baillie 2008) for a more general exploration of the effect of species composition and indicator set size on index precision.	based on their population trends between 1970 and 2006 closely matches the population dynamics of the wider community over the same time period. However, in both analyses, the magnitude of the change in our index was significantly greater because of the preferential selection of specialist species, suggesting this indicator could act as an early-warning system.
Castellón & Sieving 2012. Can Focal Species Planning for Landscape Connectivity Meet the Needs of South American Temperate Rainforest Endemics?	>Focal species< In this paper we: (1) review the body of research on landscape connectivity for understory-birds; (2) suggest some possible strategies for scaling-up corridor designs to function at larger spatial scales, by addressing population viability in addition to habitat permeability; and (3) critically evaluate the degree to which these designs may, and in many cases may not, meet the conservation needs of other vertebrates in this biome.	Used one species to develop both a fine and large scale set of conservation areas that capture connectivity as well as habitat area requirements. Intended to only represent understory birds.	Proposed habitat network is good starting point to begin stakeholder discussions. Starting point for adding other focal species, which is required to capture full set of species requirements in this biome.
Che-Castaldo & Neel 2012. Testing Surrogacy Assumptions: Can Threatened and Endangered Plants Be Grouped by Biological Similarity and Abundances?	>Umbrella management< In this study, we tested one of the fundamental assumptions underlying use of surrogate species in recovery planning: that there exist groups of threatened and	Used a comprehensive database of all plant species listed under the U.S. Endangered Species Act and tree-based random forest analysis.	We found no evidence of species groups based on a set of distributional and biological traits or by abundances and patterns of decline. Our results suggested that application of surrogate approaches for endangered

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	endangered species that are sufficiently similar to warrant similar management or recovery criteria.		species recovery would be unjustified. Thus, conservation planning focused on individual species and their patterns of decline will likely be required to recover listed species.
Cushman & Landguth 2012. Multi-taxa population connectivity in the Northern Rocky Mountains.	>Umbrella species < We evaluated the degree to which predicted connected habitat for each of 144 different hypothetical organisms expressing range of dispersal abilities and ecological responses to elevation, roads and land cover function as an indicators of connected habitat for the others in the U.S. Northern Rocky Mountains. We evaluated the effectiveness of three carnivores as connectivity umbrellas for many species.	We used resistant kernel modeling to map the extent of the study area predicted to be connected by dispersal for each species.	At relatively large dispersal abilities there was extensive overlap between connected habitat for most organisms and much of the study area is predicted to provide connected habitat for all hypothetical organisms simultaneously. In contrast, at low to medium dispersal abilities there was much less intersection of habitat connected by dispersal. We found that habitat specialists with limited dispersal ability are weak indicators of others, and likewise are weakly indicated by others. All three carnivore species performed significantly worse as connectivity umbrellas than the average across the simulated species.
Hoare et al. 2012. Can correlated population trends among forest bird species be predicted by similarity in traits?	>Management indicator< We investigated correlations among bird population trends in a mixed podocarp–hardwood forest in New Zealand in which introduced mammalian pests are controlled. We analysed trends in the abundance of 18 bird species over a 10-year period, using data from 5-min bird counts.	We used a Bayesian modelling approach to identify short-term correlations in population trends among species and to investigate whether ecological traits can be used to predict these correlated trends.	Population increases were detected in 9 of the 18 bird species over the 10-year period of the study. Population trends were correlated for 10% of species pairs (of which 81% were positive correlations). Correlations among seven of the nine species that increased in abundance were always positive; these species form a potential indicator pool. However,

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			traits were not useful for predicting correlated population trends. We advocate for testing consistency of correlations at multiple sites so as to validate the evidence-based use of the population indicator-species concept as a cost-effective alternative to monitoring whole communities.
James et al. 2012. A Methodology for Evaluating and Ranking Water Quantity Indicators in Support of Ecosystem-Based Management.	>Management indicators< This study evaluated a set of non-species management indicators in Puget Sound.	The first step in this process was the development of a general framework for selecting indicators. The framework, designed to transparently include both scientific and policy considerations into the selection and evaluation process, was developed and then utilized in the organization and determination of a preliminary set of indicators. Next, the indicators were assessed against a set of nineteen distinct criteria that describe the model characteristics of an indicator. Finally, an approach for ranking indicators was developed to explore the effects of intended purpose on indicator selection.	We identified several sets of scientifically valid and policy relevant indicators that included metrics such as annual 7 day low flow and water system reliability, which are supportive of the ecosystem approach in the Puget Sound.
Larsen et al. 2012. Birds as biodiversity surrogates: will supplementing birds with other taxa improve effectiveness?	>Surrogate for conservation areas< Birds are commonly used as surrogates of biodiversity owing to the wide availability of relevant data and their broad popular appeal. However, some studies	We explore two strategies using (i) species data for other taxa and (ii) genus- and family level data for invertebrates (when available). We used three distinct species data sets for subSaharan Africa,	We found that networks of priority areas identified on the basis of birds alone performed well in representing overall species diversity where birds were relatively speciose compared to

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	have found birds to perform relatively poorly as indicators. We therefore ask how the effectiveness of this approach can be improved by supplementing data on birds with information on other taxa.	Denmark and Uganda, which cover different spatial scales, biogeographic regions and taxa (vertebrates, invertebrates and plants).	the other taxa in the data sets. Adding species data for one taxon increased surrogate effectiveness better than adding genus- and family-level data. It became apparent that, while adding species data for other taxa increased overall effectiveness, predicting the best-performing additional taxon was difficult.
Moody & Grand 2012. Incorporating Expert Knowledge in Decision-Support Models for Avian Conservation	>Focal species< Used experts to identify focal species (broadly defined) to help identify conservation areas in the SAMBI region.	Our initial list of potential focal species comprised 65 key species identified in the SAMBI Plan. We subsequently used two processes to develop lists based on expert knowledge using two selection methods. To select focal species, we used Lambeck's (1997) selection process and a method rooted in structured decision making (SDM). Finally, we validated the two subsets of the overall list against the original list of 65 species.	Identified putative avian focal species for the region of interest. Only reduced the list somewhat (35 of 65, with only 11 species common to both methods) so monitoring requirements are still substantial.
Sutcliffe et al. 2012. Biological surrogacy in tropical seabed assemblages fails.	>"Surrogate"< A spatially and taxonomically comprehensive data set provided an opportunity for extensive testing of surrogate performance in a tropical marine system using these three approaches for the first time, as resource and data constraints were previously limiting. We measured surrogate	We defined a taxonomic group to be a surrogate for another taxonomic group if they possessed similar assemblage patterns. We investigated effects on surrogate performance of (1) grouping species by taxon at various levels of resolution, (2) selective removal of rare species from analysis, and (3) the number of clusters used to	Surrogates performed better when taxa were grouped at a phylum level, compared to taxa grouped at a finer taxonomic resolution, and were unaffected by the exclusion of spatially rare species. Mean surrogate performance increased as the number of clusters decreased. Average surrogate performance for fishes was worse overall than five other

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	performance as to how similarly sampling sites were divided into assemblages between taxa.	define assemblages, using samples for 11 phyla distributed across 1189 sites sampled from the seabed of Australia’s Great Barrier Reef. For each taxonomic group independently, we grouped sites into assemblages using Hellinger distances and medoid clustering.	taxonomic groups. Moreover, no taxonomic group was a particularly good surrogate for any other, suggesting that the use of any one (or few) group(s) for mapping seabed biodiversity patterns is imprudent; sampling several taxonomic groups appears to be essential for understanding tropical/subtropical seabed communities. Taxonomically comprehensive studies that exclude rare species would provide a better understanding of seabed assemblage patterns than studies that focus on fewer taxonomic groups and sample rare species.
Szabo et al. 2012. Adapting global biodiversity indicators to the national scale: A Red List Index for Australian birds.	>Biodiversity indicator< We present the first application of the Red List Index based on assessments of extinction risk at the national scale using IUCN’s recommended methods, evaluating trends in the status of Australian birds for 1990–2010.	We calculated Red List Indices based on the number of taxa in each Red List category and the number that changed categories between assessments in 1990, 2000 and 2010 as a result of genuine improvement or deterioration in status.	A novel comparison between trends at the species and ultrataxon (subspecies or monotypic species) level showed that these were remarkably similar, suggesting that current global red list index trends at the species level may also be a useful surrogate for tracking losses in genetic diversity at this scale, for which no global measures currently exist. The red list index for Australia is declining faster than global rates when migratory shorebirds and seabirds are included, but not when changes resulting from threats in Australia alone are considered. The index of oceanic island taxa has declined faster than

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			those on the continent or on continental islands.
Wesner & Belk 2012. Habitat relationships among biodiversity indicators and co-occurring species in a freshwater fish community.	>Biodiversity indicator< We test three hypotheses: (1) a subset of species in the overall community is useful as indicator taxa; (2) relationships between indicator species occurrence and overall fish diversity are driven by similar habitat relationships among species; (3) conservation of indicator species will positively affect co-occurring native species to shared habitat relationships among common species.	Sampled fish and habitat in multiple streams. We used multiple logistic or linear regression and model selection to identify the habitat variables associated with the occurrence and density of potential indicator species and common co-occurring native species.	Fish diversity (residual diversity and species richness) was higher at sites with potential indicators than sites without, indicating that conservation aimed at any of these species is likely to affect a broad number of co-occurring taxa. However, with minor exceptions, habitat correlations were inconsistent between indicator and co-occurring species. These data suggest that reliance on indicator taxa in conservation can be misleading because they obscure important ecological information about affected non-target species. We identified four species that are potential biodiversity indicators in our study area, but only one indicated common habitat relationships with at least 50% of co-occurring species based on both occurrence and abundance.
Carvalho et al. 2011. Incorporating evolutionary processes into conservation planning using species distribution data: a case study with the western Mediterranean herpetofauna.	>Surrogate species for defining conservation areas< To incorporate evolutionary processes into conservation planning using species distribution patterns and environmental gradients as surrogates for genetic diversity. To identify priority conservation areas, biotic elements and environmental categories were	Distributions of 154 herpetological species were predicted using maximum entropy models, and groups of significantly co-occurring species (biotic elements) were identified. Environmental gradients were characterized for the complete area and for the area covered by each biotic element, by performing a principal component	Nine biotic elements were identified - four for the amphibians and five for the reptiles. Priority areas identified in the three scenarios were similar in terms of amount of area selected, but exhibited low spatial agreement. In the present study, we identified biotic elements and environmental gradients as surrogates for the neutral and the adaptive components

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	used as surrogates for the neutral and adaptive components of genetic diversity, respectively. Priority areas for conservation were identified under three scenarios: (1) setting targets for species only; (2) setting targets for species and for each environmental category of the overall area; and (3) setting targets for each species and for each environmental category within each biotic element.	analysis on the data matrix composed of nine environmental variables. The first two principal component analysis axes were classified into four categories each, and those categories were combined with each other resulting in an environmental classification with 16 categories.	of genetic variability, respectively. Our results showed that spatial prioritization exercises that explicitly integrate such surrogates deliver quite different spatial priorities compared to plans that only account for species representation. Moreover, all solution found in each scenario showed limited agreement with the current Protected Areas network. Prioritization exercises that integrate surrogates for evolutionary processes can deliver spatial priorities that are fairly different to those where only species representation is considered.
Epps et al. 2011. An empirical evaluation of the African elephant as a focal species for connectivity planning in East Africa.	>Focal species, flagship species< We evaluated whether the African elephant, a world-recognized flagship species, would serve as an appropriate focal species for other large mammals in a potential linkage between two major protected area complexes.	We used walking transects to assess habitat, human activity and co-occurrence of elephants and 48 other large mammal species at 63 sites using animal sign and direct sightings. We repeated a subset of transects to estimate species detectability using occupancy modelling. We used logistic regression and model selection to characterize patterns of elephant occurrence and assessed correlation of elephant presence with richness of large mammals and subgroups. We considered other possible focal species, compared habitat-based linear regression models of large	Elephant presence was highly positively correlated with the richness of large mammals, as well as ungulates, carnivores, large carnivores and species > 45 kg in body mass ('megafauna'). Outside of protected areas, both mammal richness and elephant presence were negatively correlated with human population density and distance from water. Only one other potential focal species (hyaena) was more strongly correlated with species richness than elephants, but detectability was highest for elephants. Although African elephants have dispersal abilities that exceed most other terrestrial mammals, conserving

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		mammal richness and used circuit theory to examine potential connectivity spatially. Compared elephants to other species distributions, inside and outside conservation areas	elephant movement corridors may effectively preserve habitat and potential landscape linkages for other large mammal species among Tanzanian reserves.
Fattorini et al. 2011. Conserving organisms over large regions requires multi-taxa indicators: one taxon's diversity-vacant area is another taxon's diversity zone.	>Biodiversity indicator, hotspots< We analysed spatial patterns of diversity in several arthropod taxa from the Turkish fauna (Scorpiones, Chilopoda, Coleoptera Cicindelidae, Hydrophilidae (gen. Laccobius), Nitidulidae, Tenebrionidae Pimeliini, Chrysomelidae Cryptocephalinae, and Lepidoptera Hesperioidea and Papilionoidea) to test whether there are multi-group hotspots or whether different groups have different areas of maximum diversification.	We used three metrics of diversity: species richness, residuals from the species-area relationship, and species/area ratios. The study is focused on the regional scale (as opposed to global/continental or for reserve selection) .	In each group, the three metrics were significantly positively correlated. However, the hotspots identified using one metric show small agreement with those identified by other metrics. Although patterns of cross-taxon diversity were significantly and positively correlated for all metrics, hotspots of different groups show little overlap. Moreover, proportions of non-target species captured by hotspots of a target taxon were usually moderate. On the other hand, we found that hotspots of certain groups tend to be concentrated in particular regions, and some groups were good surrogates for others. For an effective conservation approach, we advocate the use of subsets of species as surrogates for all species, provided that selected subsets are representative of animals with different ecological needs and biogeographical histories.
Januchowski-Hartley et al. 2011. Coarse-filter surrogates do not represent freshwater fish	>Coarse-filter surrogate< Abiotic and biologically informed classifications are often used in	We used three agglomerative classification methods to produce biologically informed classes	Our biologically informed surrogates did not significantly improve the average representation of fish species



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diversity at a regional scale in Queensland, Australia.	conservation planning as coarse-filter surrogates for species. The relationship between these surrogates and the distribution of species is commonly assumed, but rarely assessed by planners. We derived four abiotic and eight biologically informed classifications of stream reaches to serve as surrogates for biodiversity patterns in the Wet Tropics bioregion, Queensland, Australia. We tested the effectiveness of the surrogates by calculating the average achievement of the same targets for predicted distributions of 28 fish species.	based on the relationship between the biological data for some stream reaches and abiotic data available for all streams. Our test features were the modeled occurrences of 28 freshwater fish species. We used selection-based methods to test the effectiveness of the surrogates. We identified priority areas for conservation based on the surrogates, then measured how well these areas represented the species and, finally, how well they matched the areas selected independently to represent the species.	over purely abiotic ones. Our results showed that neither abiotic nor biologically informed classifications were good at representing freshwater fish species; in fact none of the surrogates led to average representation of species better than randomly selected planning units. These results meant that selection of stream reaches to achieve surrogate targets was effectively random with respect to probabilities of fish species occurrence, leading to poor representation of fish species. We conclude there is a limited basis for using coarse-filter surrogates to represent freshwater fish diversity in this region.
Suring et al. 2011. Maintaining populations of terrestrial wildlife through land management planning: A case study.	>Focal species< We developed an 8-step process to address those species for which management for ecosystem diversity may be inadequate for providing ecological conditions capable of sustaining viable populations. Groups were based primarily on habitat associations. We selected 36 primary focal species (78% birds, 17% mammals, 5% amphibians) for application in northeast Washington State based on risk factors and ecological characteristics. We combined	We used agglomerative hierarchical cluster analysis until all species were joined in one cluster. We developed groups primarily based on commonality of vegetation type and structural stage. We also evaluated similarity between species and among clusters using the Ochiai index of similarity. We developed Bayesian Belief Network models to provide a structured tool for integrating several sources of information to make comparisons among management alternatives	Our results identified 34 focal species for northeast WA. We provide descriptions of wolverine and northern goshawks as examples of the resulting conservation planning, including management strategies and which other species are presumed to be served by them as surrogate species.

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	conservation strategies for individual species with other focal species and with management proposals for other resources (e.g., recreation, fire, and fuels management) to develop a multi-species, multi-resource management strategy.	on how well the conservation of focal species was addressed. We developed a process to prioritize focal species monitoring based on the likelihood of restoring or maintaining well-distributed, self-sustaining populations of each focal species; whether source habitat and risk factors that influenced sustainability for each species were likely to increase, decrease, or remain the same; and the degree of uncertainty.	
Trinidad-Filho & Loyola 2011. Performance and consistency of indicator groups in two biodiversity hotspots.	>Biodiversity indicators (hotspots)< Several studies have evaluated the effectiveness of indicator groups, but still little is known about the consistency in performance of these groups in different regions, which would allow their a priori selection. We systematically examined the effectiveness and the consistency of nine indicator groups in representing mammal species in two top-ranked Biodiversity Hotspots: the Brazilian Cerrado and the Atlantic Forest.	To test for group effectiveness we first found the best sets of sites able to maximize the representation of each indicator group in the hotspot and then calculated the average representation of different target species by the indicator groups in the hotspot. We considered consistent indicator groups whose representation of target species was not statistically different between hotspots.	Effective indicator groups required the selection of less than 2% of the hotspot area for representing target species. We show that several indicator groups could be applied as shortcuts for representing mammal species in the Cerrado and the Atlantic Forest to develop conservation plans, however, only restricted-range species consistently held as the most effective indicator group for such a task. This group is of particular importance in conservation planning as it captures high diversity of endemic and endangered species.
Tulloch et al. 2011. Wise selection of an indicator for monitoring the success of management actions.	>Management indicator species< Used indicator species to assess results of invasive predator control. The aim is to develop and evaluate approaches for indicator selection	Two approaches (one qualitative and one quantitative) for selecting indicators plus a case study. Started with pool of 12 mammals. Did not assess with field data	Despite being based on qualitative information from expert knowledge, when costs were incorporated in a sensible way the scoring approach assigned the same species to the top

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	in cases of limited funding and/or limited knowledge, with the objective of finding the single most cost-effective and informative indicator out of a suite of potential species.	going forward, only retrospective analysis of long-term data set - detailed methods in paper	rank as the quantitative metric, suggesting that a qualitative approach that accounts for the costs of monitoring has the potential to prioritise the same species as a quantitative approach based on empirical data. The new quantitative cost-effectiveness approach developed here will allow transparent, explicit, credible, accountable selection of indicator species that can demonstrate improved performance of environmental management programmes and show that money has been effectively used to produce environmental benefits.
Banks et al. 2010. The Use of Surrogate Species in Risk Assessment: Using Life History Data to Safeguard Against False Negatives.	>Environmental indicator species < Evaluated potential surrogate fish species for listed salmon relative to toxicant exposure.	Using fecundity assessments, life history trait comparisons, and modeling to compare and predict which species may be best surrogate species. Modelled fecundity reductions for all species and assessed how well the surrogate species fecundity reductions matched the listed species.	Generally, our results serve as a cautionary tale for relying solely on a few surrogate species to represent endangered/threatened species across a range of taxa. For most endangered/threatened wildlife, extrapolations from surrogate species' responses to disturbance seem woefully simplistic in the context of more complex ecological factors that influence their population dynamics.
Barua et al. 2010. Mutiny or Clear Sailing? Examining the Role of the Asian Elephant as a Flagship Species.	>Flagship species< Evaluated flagship species to further landscape conservation in India.	Used durveys of people's attitudes toward elephant conservation.	Survey results showed that exposure to wild elephants negatively affected intentions to conserve elephants, while specific concern for the elephant and direct involvement in conservation activities led to positive

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			intentions. These results suggest that the effective use of the Asian elephant as a flagship may be contingent on mitigating human–elephant conflict.
Cushman et al. 2010. Use of Abundance of One Species as a Surrogate for Abundance of Others.	>Indicator species< The central objective was to identify individual bird species that effectively indicated the abundance of other species or groups of species and to determine if species grouping strategy and spatial scale of analysis influenced the apparent strength and number of surrogate relationships.	Results of analyses based on presence–absence or relative abundance across all plots or only plots located in patch interiors produced were extremely similar and provided identical interpretation of the number, strength, and nature of surrogate relationships. So results focus on relative abundance across all plots. We evaluated patterns among species across sample plots and we evaluated a second scale of co-occurrence on the basis of similarity of abundances within each of the 30 sub basins.	Surrogacy strength at the sub basin level was not substantially different than at the plot level for most species groups. Empirical grouping provided stronger patterns of surrogacy than the a priori grouping. Even here, however, little variance was explained. The absence of significant patterns of co-occurrence for any of the other groups suggests that coarse habitat-association attributes may be the only characteristics useful for defining species surrogacy. Guild-level indicator species for birds are not effective in this system.
Grantham et al. 2010. Effectiveness of Biodiversity Surrogates for Conservation Planning: Different Measures of Effectiveness Generate a Kaleidoscope of Variation.	>Environmental surrogates< We identified four factors likely to have a strong influence on the apparent effectiveness of surrogates: (1) the choice of surrogate; (2) differences among study regions, which might be large and unquantified (3) the test method, that is, how effectiveness is quantified, and (4) the test features that the surrogates are intended to represent. Analysis of an unusually rich dataset enabled	Four methods tested the effectiveness of the surrogates by selecting areas for conservation of the surrogates then estimating how effective those areas were at representing test features. One method measured the spatial match between conservation priorities for surrogates and test features. For methods that selected conservation areas, we measured effectiveness using two analytical approaches: (1) when	In general, the effectiveness of surrogates for our taxa (mostly threatened species) was low, although environmental units tended to be more effective than forest ecosystems. The surrogates were most effective for plants and mammals and least effective for frogs and reptiles. The five testing methods differed in their rankings of effectiveness of the two surrogates in relation to different groups of test features. There were differences between study areas in

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	us, for the first time, to disentangle these factors and to compare their individual and interacting influences. Using two data-rich regions, we estimated effectiveness using five alternative methods (two forms of incidental representation, two forms of species accumulation index and irreplaceability correlation) to assess the performance of ‘forest ecosystems’ and ‘environmental units’ as surrogates for six groups of threatened species.	representation targets for the surrogates were achieved (incidental representation), or (2) progressively as areas were selected (species accumulation index).	terms of the effectiveness of surrogates for different test feature groups. Overall, the effectiveness of the surrogates was sensitive to all four factors. This indicates the need for caution in generalizing surrogacy tests.
Moseley et al. 2010. A Multi-Criteria Decisionmaking Approach for Management Indicator Species Selection on the Monongahela National Forest, West Virginia.	>Management indicator species< We used the analytical hierarchy process (AHP) to determine the best management indicator species (MIS) for three management objectives of the 364,225-ha Monongahela National Forest (MNF) in West Virginia.	We compiled a set of alternative MIS, including current MNF MIS, for each objective based on a literature review of species-habitat relations in the Appalachian Mountain region. We used the AHP to determine local priorities, based on pair-wise comparisons for criteria and MIS alternatives. Among potential alternatives, total global priority scores for the ruffed grouse, pileated woodpecker, and Virginia northern flying squirrel contributed most to respective management objectives.	We believe the AHP is an effective tool for MIS selection, particularly within complex Appalachian ecosystems, because it provides a formal structured decision procedure, has a strong theoretical foundation, accommodates incomplete ecological data, and offers transparency to the MIS decisionmaking process.

## APPENDIX E: CASE STUDY

### FLORIDA'S WILDLIFE HABITAT CONSERVATION SYSTEM

#### Summary

As part of an effort to identify state conservation areas needed to preserve habitat for particular species, the Florida Game and Fresh Water Fish Commission (now Florida Fish and Wildlife Conservation Commission) used a combination of focal species (exclusively terrestrial vertebrates), rare plants and rare communities to identify Strategic Habitat Conservation Areas (SHCAs). “Focal species” as used in this process does not meet the definition presented in Lambeck (1997) but generally indicate priority species. These priority species, however, do serve as surrogate species during parts of this process. These areas could be protected by acquisition, easements or other means. This was summarized in a 1994 report entitled *Closing the Gaps in Florida's Wildlife Habitat Conservation System* (Cox et al. 1994). These SHCAs were identified to assist state, local and non-profit entities in focusing their land conservation and acquisition efforts.

The Commission identified a set of 44 focal species to serve as “umbrella” or “indicator” species of biological diversity in Florida and assembled as much information as they could find on the locations of these species. They assessed the level of security provided the focal species by the current system of conservation areas and proposed SHCAs for species lacking adequate representation in current conservation areas. They considered how protection of an area for a single species might also protect larger communities and multi-species assemblages; therefore these focal species were used as surrogate species to represent the larger biological communities. They also assembled information on the locations of other key components of biological diversity, including rare plants, invertebrates, and natural communities, and used that to identify additional SHCAs.

The results of this effort were analyzed more than a decade later to test the surrogate species approach: how well existing conservation lands the SHCAs from 1994 would protect 124 rare wildlife species (Cox & Kautz 2000). In other words, they assessed how well these focal species served as surrogates for other beneficiary species. Taken together with 54 focal species originally addressed in 1994, the 124 species include all species of vertebrates (except fishes, marine mammals, and sea turtles) listed as endangered, threatened, or species of special concern; species believed to be in some degree of imperilment by the Florida Committee on Rare and Endangered Plants and Animals; and wildlife identified by other experts (e.g., Millsap et al. 1990).

This was followed by a more comprehensive update to the identified SHCAs, the results of which were summarized in a report entitled *Wildlife Habitat Conservation Needs in Florida: Updated Recommendations for Strategic Habitat Conservation Areas* (Endries et al. 2009). This update incorporated increased public ownership and easements, changes in land use, updated and more detailed source data (i.e., land cover maps, species occurrences, etc.), and improvements in population viability modeling. This update modified the focal species list, but kept even more strictly to the limitation of only terrestrial vertebrate species. They did not explicitly include rare plants or rare communities.

### Overall Comparison

The species analyzed in 2009 include all species for which SHCA were identified in *Closing the Gaps*, species whose habitats were not adequately protected in the *Habitat Conservation Needs of Rare and Imperiled Wildlife Report* (Cox & Kautz 2000), species subject to changes in their federal listing status (recently completed and proposed), and additional species whose habitats are thought to be threatened because of recent population trends. Furthermore, rare plants and natural-community types were not directly assessed in 2009, as they were in 1994. The 2009 study was also limited exclusively to terrestrial vertebrate species. See below under Focal Species Selection for a comparison of the species used.

Construction of a potential-habitat map in 2009 did not necessarily follow the same steps used to create the SHCA map in 1994, although the general process was similar. There was much more detailed data and new data sets available in 2009 and there had been significant improvements to population viability modeling.

The 1994 study accomplished population viability modeling principally by identifying sufficient potential habitat to sustain a minimum of 10 populations of 200 individuals. The 2009 study determined the probability of a population declining to a certain level within a given time period. In most cases, they identified areas that could support the population abundance target for each species by using PVA techniques to determine the optimal population size and persistence set for each species in the study.

Areas identified as SHCA in 2009 are substantially different from those identified in 1994. In the current study, more land is identified as SHCA, the type and area of land-cover classes identified as SHCA are different, and the area and distribution of SHCA within counties are different. This is expected given the differences in data, techniques, and procedures used in the two projects.

The number of individual species identified as needing SHCA was similar between the two studies, 30 species in the 1994 study and 34 species in the 2009 study. There are 21 species

that were evaluated in both studies and were found to need SHCA in 1994 and 2009. However, important differences emerge in the species that were evaluated and the corresponding results between the two studies. SHCAs were created for 9 species in 1994 that were deemed unnecessary in 2009 based on PVA results. This difference may be the result of public land additions between 1994 and 2009. Additionally, a better understanding of the life history of these species and more location data may have contributed to the difference. Two species evaluated in 1994 did not warrant SHCA at the time, but they did warrant SHCA in the 2009 evaluation. This part of the process did not assess differences in effectiveness to serve as surrogates, but were focused on the focal species only.

*Based on the 2009 analyses, even if all potential habitat was placed under conservation protection, most of the species still did not meet the minimum population persistence goals and will face threats of continued population decline or extinction. This is an alarming phenomenon and one that warrants additional research to identify whether Florida truly has reached a threshold of increasing species extinctions or declines in abundance caused by existing and continued habitat loss and fragmentation.*

Due to the acreage identified as SHCA in 2009, they prioritized the SHCA to highlight those areas in need of more immediate protection while still recognizing the habitat protection needs of all the species with SHCA. Species with more serious risk of extinction were given a higher priority than those with less risk of extinction, based on global and state ranks of rarity. The surrogacy value of these species was not a factor in this prioritization.

### Goals and Objectives

The goal of the 1994 project was to identify lands in Florida that, at a minimum, must be conserved and managed in order to ensure the long term survival of key components of Florida's biological diversity. To achieve this goal, the analysis included the following:

- 1) Identify a set of 44 focal species to serve as "umbrella" or "indicator" species of biological diversity in Florida and assembled as much information as possible on the locations of these key species.
- 2) Assess the level of security provided these focal species by the current system of conservation areas, and propose SHCAs for 30 species lacking adequate representation in current conservation areas. The proposed SHCAs for each species were based on the most recent information available on conservation area planning and factoring how a SHCA for a single species might also protect larger communities and multispecies assemblages. The results were weighted somewhat based on the supposed surrogacy value of a given focal species.



- 3) In addition to the 44 focal species, assemble as much information as possible on the locations of other key components of biological diversity, including rare plants, invertebrates, and natural communities to identify additional SHCAs.
- 4) Develop regional maps displaying information on the distribution of rare plants, animals, and natural communities (i.e., hotspots).

The goal of the 2009 project was to identify the minimum amount of land needed in Florida to ensure the long term survival of key components to Florida's biological diversity. The objectives set in 2009 to reach that goal were:

- 1) Select the species that compose the focal group for analysis.
- 2) Produce a map of potential habitat for each species.
- 3) Determine if the amount of potential habitat that exists on lands managed for conservation is adequate for the long-term persistence of the all species in Florida (assuming the focal species are operating as surrogate species for the rest of the terrestrial vertebrate communities).
- 4) If warranted, identify suitable privately owned lands in the state that would benefit the long-term persistence of the species in Florida.
- 5) Make any additional recommendations regarding research, management, and habitat protection relevant to ensuring the species' long-term viability.

#### *Focal Species Selection*

Of the 542 taxa of terrestrial vertebrates listed by Millsap et al. (1990) as occurring regularly in Florida, 44 were selected for in-depth analyses in 1994. Another 120 vertebrate taxa were analyzed either as part of multi-species assemblages or as part of a "gap" analysis. The 44 "focal species" were selected based on their utility as indicators of natural communities (how this was determined was not specified in the report) or because they require suitable habitat conditions covering large areas. Statewide habitat and distribution maps were created for each of the 44 focal species using data on known locations of occurrence, information on the land cover and vegetation types used by each species, and published or well documented information on the life-history requirements of the species.

Several criteria were used to select focal species in 1994. A primary consideration was whether habitat requirements for the species could be described using the land-cover map and other geographic data sets. A second consideration was whether a species exhibited large home-range requirements and might be susceptible to increasing fragmentation of contiguous forest tracts. A third consideration was whether a species was closely tied to a specific rare plant community so that conservation plans for a focal species might provide greater protection for rare communities (i.e., function as a surrogate for a larger group of species). A final group of birds was also included as focal species because they are listed as endangered or threatened in Florida, exhibited declining populations or special habitat

requirements, or, most importantly, were the subject of special studies that resulted in precise data on known occurrences. The relatively larger proportion of birds chosen as focal species reflects a greater knowledge of the distributions and habitat requirements of this group.

In addition to the overall SHCA identification in 1994, the project also identified regional biodiversity hotspots using focal species. They constructed “hot spot” maps of biological resources for each region by overlaying the habitat maps developed for the 44 focal taxa, wading birds, and important natural communities and subdividing the composite map into three broader categories of Class 1, Class 2, and Class 3 areas based on the number of focal species that would likely find appropriate habitat conditions in the area. Class 1 lands depict areas where habitat conditions for 3-4 focal species likely occur; Class 2 lands show areas where habitat conditions for 5-6 focal species likely occur; and Class 3 lands show areas where habitat conditions for 7+ focal species likely occur. Class 1 lands are often large forested tracts that have varying degrees of natural quality. These were created on the assumption that identifying the hot spot in each region would allow local governments and non-profits to implement conservation measures that would benefit all the species in those areas (i.e., areas of high overlap of the focal species also indicated high overlap of all species).

In 2009, they included both community indicators and umbrella species (with some keystone species serving as umbrella species), although they did not describe how they determine whether a species played one of these roles. Examples of community indicators included were Florida scrub-jay and crested caracara. Gopher tortoise (*Gopherus polyphemus*) are a keystone species for which more than 300 other species (e.g., indigo snake, gopher frog) use their burrows (Diemer 1992) and were used as an umbrella species in 2009. Examples of other umbrella species used in 2009 include the Florida black bear, Cooper’s hawk, and Florida panther.

Selection of the focal species for the 2009 study was accomplished by:

- including species for which SHCA were identified in 1994, but incorporating changes in potential habitat, newly protected habitat, and new information on the species;
- including the 17 species whose habitats were determined to be inadequately protected in Cox & Kautz (2000); and
- including additional focal species identified by the Commission biologists as having declining populations and threatened habitats; for which new information would alter their habitat and SHCA analysis; and species proposed for change in listing as endangered, threatened, or species of special concern.

TABLE E-8. FOCAL SPECIES COMPARISON BETWEEN 1994 AND 2009 REPORTS FOR CREATING SHCAs (FROM ENDRIES ET AL. 2009). SPECIES IN BOLD ARE UNIQUE TO EACH STUDY.

1994	2009
American crocodile	American crocodile
Anastasia Island beach mouse	Anastasia Island beach mouse
Atlantic salt marsh snake	Atlantic salt marsh snake
<b>Black-whiskered Vireo</b> <sup>1</sup>	Big Cypress fox squirrel
Big Cypress fox squirrel	Choctawhatchee beach mouse
<b>Bog frog</b> <sup>1</sup>	<b>Cooper's Hawk</b> <sup>2</sup>
Choctawhatchee beach mouse	Cuban Snowy Plover
<b>Crested Caracara</b> <sup>1</sup>	Florida black bear
Cuban Snowy Plover	<b>Florida Burrowing Owl</b>
Florida black bear	Florida Grasshopper Sparrow
Florida Grasshopper Sparrow	<b>Florida mouse</b> <sup>2</sup>
Florida panther	Florida panther
<b>Florida Sandhill Crane</b> <sup>1</sup>	<b>Florida salt marsh vole</b> <sup>2</sup>
Florida Scrub-Jay	Florida Scrub-Jay
Gulf salt marsh snake	<b>Gray bat</b> <sup>2</sup>
<b>Limpkin</b> <sup>1</sup>	Gulf salt marsh snake
Louisiana Seaside Sparrow	<b>Key deer</b> <sup>2</sup>
Mangrove Cuckoo	Louisiana Seaside Sparrow
<b>Mottled Duck</b> <sup>1</sup>	<b>Lower Keys marsh rabbit</b> <sup>2</sup>
<b>Red-cockaded Woodpecker</b> <sup>1</sup>	Mangrove Cuckoo
Scott's Seaside Sparrow	<b>Pine barrens tree frog</b>
Short-tailed Hawk	<b>Sand skink</b> <sup>2</sup>
MacGillivray's Seaside Sparrow	<b>Sanibel Island rice rat</b> <sup>2</sup>
Snail Kite	Scott's Seaside Sparrow
<b>Southeastern American Kestrel</b> <sup>1</sup>	<b>Seal salamander</b> <sup>2</sup>
Southeastern beach mouse	Short-tailed Hawk
<b>Southern Bald Eagle</b> <sup>1</sup>	<b>Silver rice rat</b> <sup>2</sup>
St. Andrews beach mouse	MacGillivray's Seaside Sparrow
Swallow-tailed Kite	Snail Kite
White-crowned Pigeon	Southeastern beach mouse
	St. Andrews beach mouse
	<b>Striped newt</b> <sup>2</sup>
	Swallow-tailed Kite
	White-crowned Pigeon

<sup>1</sup> Species did not require SHCA in 2009. <sup>2</sup>Species not evaluated in 1994.

### Outcomes from the 1994 Report

As of May 2009, 4.5 million ha of land were set aside for some type of conservation use, while only 2.81 million ha were protected in 1994. The 1994 study identified 1.9 million ha of privately owned lands as SHCA. Since 1994, 0.59 million ha of lands mapped as SHCA for biodiversity conservation have come under public protection. The 2009 study identified 3.6 million ha as SHCA, even after accounting for the 0.59 million ha now under public protection.

If all lands identified as SHCA in 2009 were combined with the 4.5 million ha then under public management, nearly 8.1 million ha (53% of the total non-water area of Florida) would be under some form of protection for species and habitat conservation. All 67 Florida

counties experienced a change in the total area identified as SHCA from 1994 to 2009. Most counties had more area identified as SHCA in 2009 with the mean change being 13.88%. When a county experienced a large net loss in SHCA, it was usually due to large land parcel acquisitions since the 1994 report.

Based on the results from Cox & Kautz (2000), the surrogate species approach taken in 1994 worked quite well in its attempts to identify important habitats for rare and declining species in Florida: Of the 124 species evaluated, only 17 were found to need additional habitat conservation efforts. In other words, the use of the focal species as surrogates for at least these 124 other species basically worked. Those lacked the minimum habitat to support 10 populations of 200 breeding adults on existing conservation lands and SHCAs. The habitats identified to meet the conservation needs of the 17 species encompass <5% of the SHCAs identified in Cox et al. (1994) and do not consist of very large blocks of habitat, principally because most of the species considered here have small home ranges and restricted distributions. Thus, the long-term survival for most of these species could be secured through protection efforts aimed at very small parcels of land, and, as a consequence, no new SHCAs were identified as a result of this work.

A few examples of how the concepts and information presented by Cox et al. (1994) have been used include the following:

- The number of acres of SHCA purchased was specified in statute as a measurable goal for Florida Forever (Florida Statutes 259.105).
- Lands identified in *Closing the Gaps* were used to evaluate and rank proposals submitted to Preservation 2000 and Florida Forever land-acquisition programs.
- The Florida Communities Trust land-acquisition program used the presence of SHCA on a prospective parcel of land as a ranking criterion for eligibility to receive funding.
- SHCA were one of the data layers used to identify and rank lands for acquisition as part of the Florida Forever Conservation Needs Assessment (FNAI 2000).
- SHCA were one of the layers used by the University of Florida GeoPlan Center to identify ecological greenways (Hector et al. 2000).
- Several of Florida's 11 regional planning councils incorporated SHCA into maps of natural resources of regional significance as part of the process for developing Strategic Regional Policy
- Plans prescribed by Florida law in 1995 (Florida Statutes 186).
- SHCA and biodiversity hot spots identified in *Closing the Gaps* were considered "best available data" used by local governments as part of Evaluation and Appraisal Reports required to update comprehensive land-use plans on a five-year cycle.
- SHCA were used as an input to rank the Florida landscape with respect to their importance to wildlife as part of the the Commission's Integrated Wildlife Habitat

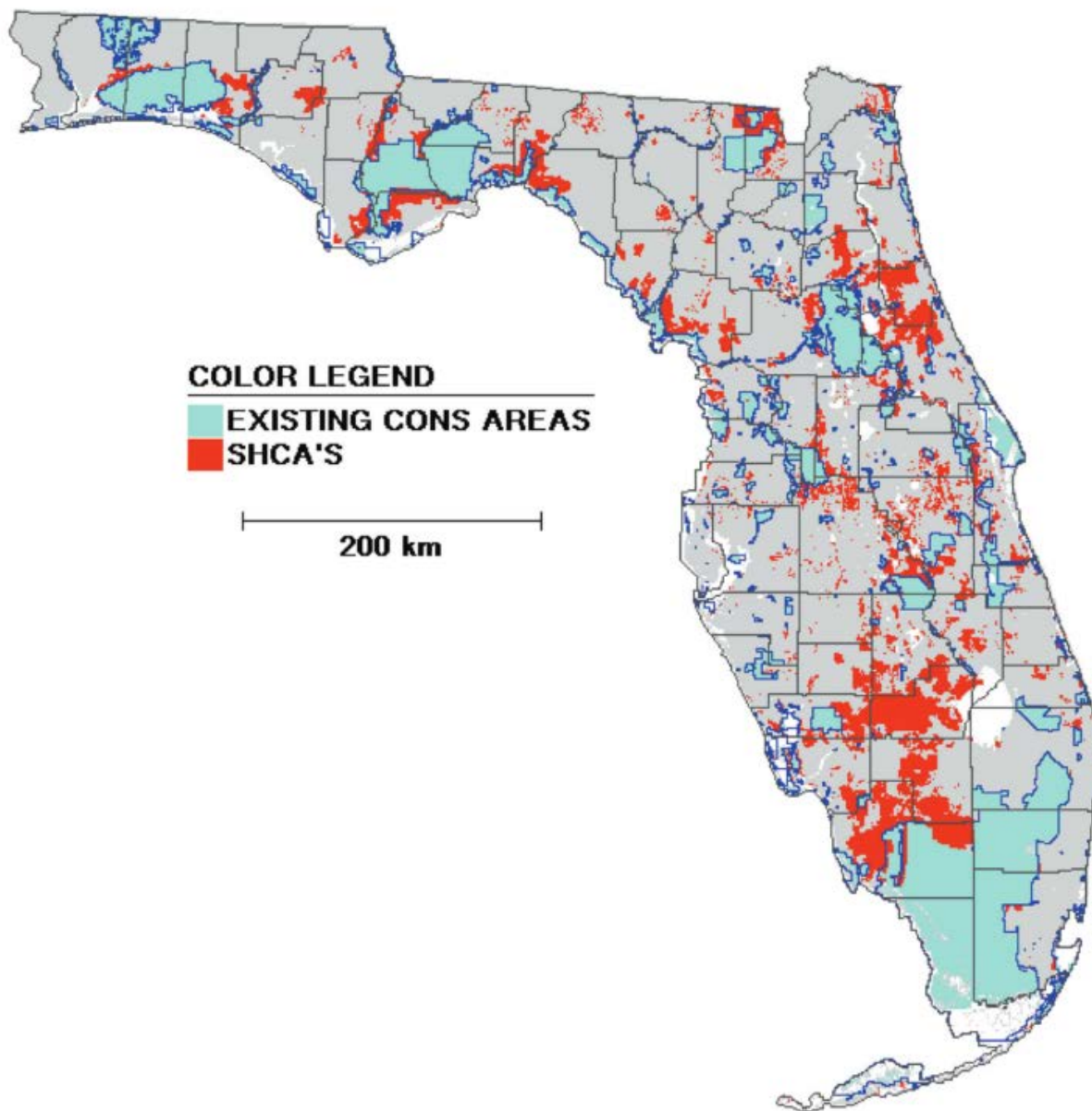
Ranking System (2001) data set (Endries et al. 2003), which was produced at the request of the FDOT as a tool for rapidly evaluating the likelihood that new road projects would adversely affect important wildlife areas.

- Chapter 373 (Part IV) (Florida Statutes) and Chapter 40 (Florida Administrative Code), which prescribe procedures to be followed to obtain an Environmental Resource Permit, specify that impacts to fish and wildlife must be considered, and data in *Closing the Gaps* have been used for this purpose.
- Rule 9J-5 (Florida Administrative Code) requires that assessments of effects on wildlife habitats resulting from Developments of Regional Impact must be made using “best available data,” which includes SHCA and biodiversity hot spots from *Closing the Gaps*.
- *Closing the Gaps* has been translated into Japanese by the Ecosystem Conservation Society of Japan and used as a model approach for conservation planning in Japan.
- *Closing the Gaps* was used as the example of how to conduct regional conservation planning by a team of scientists that developed a set of measurable objectives for application to conservation planning efforts (Tear et al. 2005).

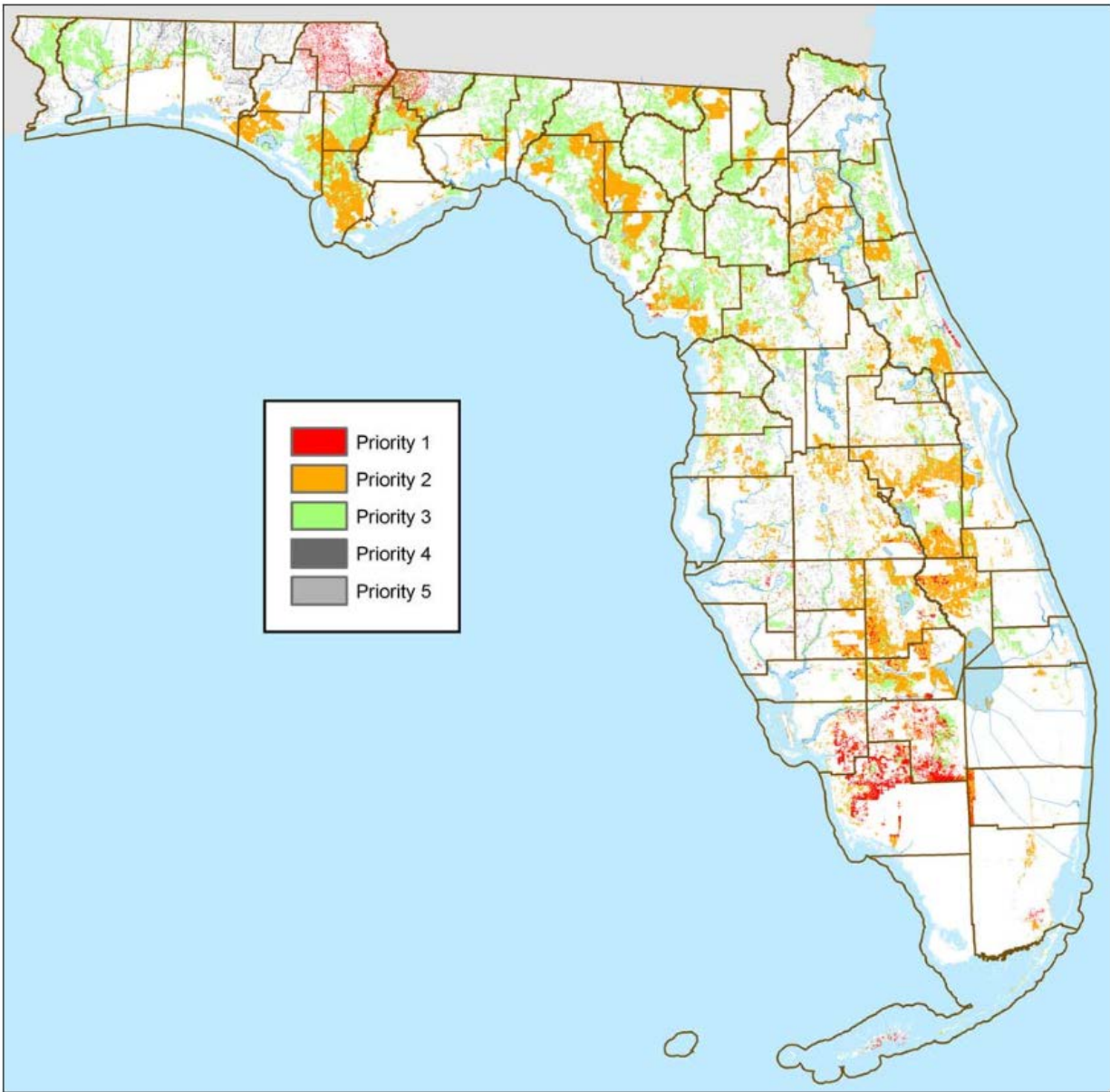
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**FIGURE E-1. SHCA'S IDENTIFIED IN THE 1994 REPORT (FROM COX ET AL. 1994).**



**FIGURE E-2. SHCAs IDENTIFIED IN 2009, INCLUDING THEIR PRIORITY WHICH WAS DETERMINED BY THE SPECIES PROTECTED (FROM ENDRIES ET AL. 2009).**