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A Framework to Integrate Habitat Monitoring and Restoration with Endangered Insect Recovery

Jason Bried · Tim Tear · Rebecca Shirer · Chris Zimmerman · Neil Gifford · Steve Campbell · Kathy O'Brien

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Abstract Monitoring is essential to track the long-term recovery of endangered species. Greater emphasis on habitat monitoring is especially important for taxa whose populations may be difficult to quantify (e.g., insects) or when true recovery (delisting) requires continuous species-specific habitat management. In this paper, we outline and implement a standardized framework to facilitate the integration of habitat monitoring with species recovery efforts. The framework has five parts: (1) identify appropriate sample units, (2) select measurable indicators of habitat requirements, (3) determine rating categories for these indicators, (4) design and implement appropriate data collection protocols, and (5) synthesize the ratings into an overall measure of habitat potential. Following these steps, we developed a set of recovery criteria to estimate habitat potential and initially assess restoration activities in the context of recovering an endangered insect, the Karner blue butterfly (*Lycaeides melissa samuelis*). We recommend basing the habitat potential grading scheme on recovery plan criteria, the latest information on species biology, and working hypotheses as needed. The habitat-based assessment framework helps to identify which recovery areas and habitat patches are worth investing in and what type of site-specific restoration work is needed. We propose that the transparency and decision-making process in endangered insect recovery efforts could be improved through adaptive management that explicitly identifies and tracks progress toward habitat objectives and ultimate population recovery.

Keywords Endangered species · Karner blue butterfly · Monitoring and evaluation · Recovery planning · Restoration monitoring

J. Bried (✉)
Department of Zoology, Oklahoma State University, 501 Life Sciences West, Stillwater, OK 74078, USA
e-mail: bried@okstate.edu

T. Tear
Grumeti Fund, P.O. Box 0, Mawalla Park, Olasiti Road, Arusha, Tanzania

R. Shirer · C. Zimmerman
The Nature Conservancy, Eastern New York Chapter, 195 New Karner Road, Suite 201, Albany, NY 12205, USA

N. Gifford · S. Campbell
Albany Pine Bush Preserve Commission, 195 New Karner Road, Suite 1, Albany, NY 12205, USA

K. O'Brien
Endangered Species Unit, New York State Department of Environmental Conservation, 625 Broadway, Albany, NY 12205, USA

Introduction

Monitoring and evaluation are relatively neglected aspects of conservation science (Stem et al. 2005; Lindenmayer et al. 2011). The deficit is prominent for invertebrates (Rohr et al. 2007), and especially problematic when legal, financial, or biological stakes are high, such as found in endangered species recovery (Campbell et al. 2002). The dearth of monitoring for endangered invertebrates may be linked to the logistical difficulties inherent to quantifying populations of hyper-diverse taxa (Rohr et al. 2007), and to the false perception that monitoring yields too little useful information at too much cost (Lovett et al. 2007).

Campbell et al. (2002) emphasized a need to expand beyond population-based metrics in endangered species recovery planning. More recently, Neel et al. (2012)

showed that attention to habitat is severely lacking in U.S. recovery efforts and explicitly called for establishing habitat assessment criteria to improve the recovery process. Increasing the focus on habitat is consistent with the U.S. Endangered Species Act directive to conserve "the ecosystems upon which species depend" (ESA Sec 2(b)), and supports the mandate to determine the portion of the range where endangerment occurs (ESA Sec 3(6) and 3(20)).

Habitat monitoring is particularly important when recovery involves species-specific habitat restoration (Scott et al. 2005). Restoration is typically expensive, labor-intensive, and can cause drastic and potentially unexpected changes in habitat conditions. As such, restoration practices require documentation of positive outcomes to justify the significant effort and expense (Ruiz-Jaen and Aide 2005; Zedler 2007). Despite a recent increase of empirical papers tracking restoration progress (Wortley et al. 2013), there is still an overall paucity of these studies. For example, despite the significant investment of >$1 billion annually in river restoration across the United States, <10 % of the projects documented any results, and few projects tracked their progress or reported on positive or negative findings, limiting opportunities to learn and improve (Bernhardt et al. 2005). Similarly, a systematic review of restoration efforts for at-risk butterflies in the U.S. and Britain highlighted that a majority of species require and receive active habitat restoration, but only a few track the ecological responses of those activities (Schultz et al. 2008).

A habitat-based approach may prove beneficial in the context of endangered insect recovery because insect diversity is expected to parallel vegetation trajectories and spatial patterns (Dennis et al. 1998; Panzer and Schwartz 1998). Consequently, attention toward habitat provides not only direct assessment of restoration activities but may also offer mechanistic and predictive insight for how populations and communities will respond to those activities. In this view, habitat characteristics are leading proximate indicators of potential species recovery (Bried 2009). Habitat changes may provide indirect evidence of recovery progress, and should be most beneficial when population- or community-level estimates are difficult to obtain, which is true of many insects.

Conservation planners and managers require a standardized framework to facilitate the integration of habitat monitoring with recovery efforts for insects. In this paper, we outline and implement a specific framework for establishing indicators of adaptively-managed conservation projects (Parrish et al. 2003) and synthesize the information into scale-appropriate estimates of habitat potential using associated software incorporated into the Open Standards for Conservation (Schwartz et al. 2012). We demonstrate the framework for the endangered Karner blue butterfly (KBB) (*Lycaeides melissa samuelis*), an example where recovery depends on restoring individual habitat patches and biologically appropriate habitat networks (Summerville et al. 2005; McIntyre et al. 2007; Pickens and Root 2009). This paper focuses on habitat potential rather than habitat quality because we do not include population-based performance measures (occurrence, abundance, fitness, survival, productivity) for the target species (e.g., McIntyre et al. 2007; Suhonen et al. 2010).

Study System

The historic range of the KBB spanned from Minnesota to Maine, including southern Ontario (Fig. 1). Over the past 50–55 years, the KBB has been extirpated from at least half a dozen states and all of Canada. With a few notable exceptions in Wisconsin and Michigan, all extant populations are small and prone to extinction (USFWS 2003). Throughout the range, habitat has been lost to human development and degraded by alteration of natural disturbance regimes (Smallidge and Leopold 1997). Representative KBB habitat is limited to remnant oak savannas and open-canopy pine barrens along with anthropogenic sites such as airfields, sandpits, and transmission line corridors (Andow et al. 1994; Smallidge et al. 1996; Forrester et al. 2005). These areas must contain sufficient populations of *Lupinus perennis* (hereafter lupine) because KBB larvae feed exclusively on above-ground vegetative parts of lupine. Adults, however, have broad nectar diets including over a dozen priority forage species, such as *Asclepias tuberosa*, *Ceanothus americanus*, *Lespedeza capitata*, and *Tephrosia virginiana*.

Like many at-risk Lepidoptera historically associated with open shrublands and prairie-like habitats of the region (Wagner et al. 2003), the KBB requires active management to restore and maintain suitable habitat (Smallidge et al. 1996; Kwilosz and Knutson 1999; King 2003; Pickens and Root 2008; Pavlovic and Grundel 2009). Current management practices for KBB use mowing and prescribed fire to arrest succession and plantings to increase larval and adult food availability. Mechanical and chemical treatments are employed to create new habitat patches and control woody plant encroachment that limits butterfly activity and food plant abundance. Maintaining abundant lupine is a major conservation objective across the KBB range (USFWS 2003), but restoration efforts also focus on removing invasive species, improving nectar resources, and increasing habitat heterogeneity to better accommodate the multiple factors that may affect KBB behavior and population dynamics (Grundel and Pavlovic 2007; Pickens and Root 2009).

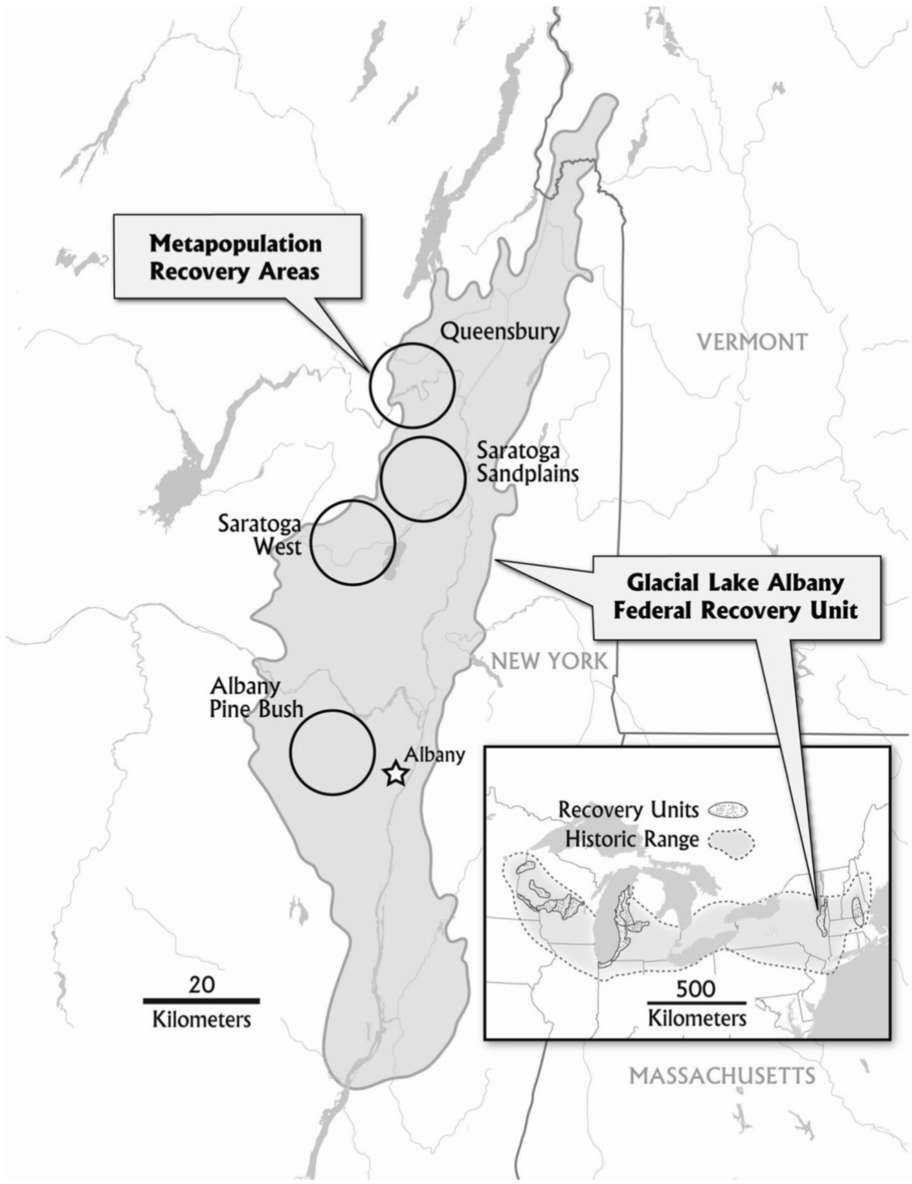

Fig. 1 The New York recovery unit (Glacial Lake Albany) and approximate locations of metapopulation recovery areas, along with current and historic distribution of the federally endangered Karner blue butterfly. Data sources: federal recovery plan (USFWS 2003) and New York State Department of Environmental Conservation

Our study is focused in the Glacial Lake Albany recovery unit in New York (Fig. 1). This unit supports the remaining naturally occurring populations of KBB on the eastern end of its range. The Glacial Lake Albany sand plain (3,750 km^2) is characterized by glacio-lacustrine deposits, gently rolling topography, and a cold-temperate humid climate. Additional details of the Glacial Lake Albany landscape and KBB recovery program in New York are documented through multiple grant reports available upon request.

Habitat Assessment Framework

Here we describe a framework for assessing habitat potential in the context of endangered insect recovery, presenting specific rationale and methodology for the KBB as an example. The framework requires that we (1) identify appropriate sample units, (2) select measurable indicators of habitat requirements, (3) determine rating categories for these indicators, (4) design and implement appropriate data collection protocols, and (5) synthesize the ratings into an overall measure of habitat potential.

Units of Observation

Habitat patches (i.e., open to partial-canopy areas containing lupine; populations may be naturally occurring or planted during restoration) were used as the unit of observation for this study because they delineate the boundaries of restoration activities and the KBB population surveys. We defined a subpopulation as including all

Table 1 Habitat requirement indicators, methods, and rating scheme used to estimate patch-scale habitat potential for the KBB in the Glacial Lake Albany federal recovery unit (New York State, USA)

Indicator	Assessment method	Indicator rating			
		Poor	Fair	Good	Very Good
Lupine density (stems per hectare)	$\sum_{i=1}^{N}(l/t)_i/N$ where a transect (i) of area t (in ha) contains l observed lupine stems, and N is the total number of lupine transects for the habitat patch	≤4,450	4,451–5,930	5,931–8,900	>8,900
Spring nectar species richness	Number of priority nectar species (total for patch) that flower during first brood flight period of KBB	0	1	2–3	≥4
Summer nectar species richness	Number of priority nectar species (total for patch) that flower during second brood flight period of KBB	0	1	2–4	≥5
Nectar mean frequency (percent quartiles)	$\sum_{i=1}^{N}(f/f_{\max})_i/N$ where f is number of subplots (4 m^2 and spaced 2 m apart) occupied by priority nectar species and $f_{\max}$ is number of subplots (usually 8) on transect i, and N is total number of nectar transects for the habitat patch	≤25	25.1–50	50.1–75	>75
Nectar evenness (index)	$\left(-\sum\frac{n_i}{N}\times\log_{10}\frac{n_i}{N}\right)\div\log_{10}S_{\text{obs}}$ where n_i/N = the proportional frequency of the ith nectar species, and S_{obs} = total number of observed nectar species for the patch (i.e., spring + summer richness)	≤25	25.1–50	50.1–75	>75
Grass cover (%)[a]	$\sum_{i=1}^{N}(g/g_{\max})_i/N$ where g = number of native grass point-intercept hits and $g_{\max}$ = number of meters (usually 30) on transect i, and N = total number of structure transects for the habitat patch	<5 or >95	5–20 or 71–95	21–30 or 51–70	31–50
Overstory cover (%)[b]	$\sum_{i=1}^{N}(o/o_{\max})_i/N$ where o = number of overstory point-intercept hits and $o_{\max}$ = number of meters (usually 30) on transect i, and N = total number of structure transects for the habitat patch	<5 or >50	31–50	16–30	5–15
Shade heterogeneity	Percent of total transects with > 30 % density of shade (trees or shrubs > 2 m tall)	≤5, >80	5.1–20 or 60.1–80	20.1–60	20.1–60[c]

[a] Based on three dense-foliage native species: *A. gerardii*, *S. scoparium*, and *S. nutans*

[b] Woody overhead canopy (>2 m) measured by point-intercept sampling and periscope densitometer

[c] Trees and shrubs each contribute at least 5 % of the cover; this contrasts with the good rating in which shade density can be comprised entirely of trees or entirely of shrubs

patches within 200 m of an occupied patch, based on the lower end of average net (lifetime) movement of KBB (Knutson et al. 1999; USFWS 2003). A patch is considered occupied when KBB adults have been observed at least once within a 5-year time frame.

Habitat potential was assessed for patches within each of the four identified recovery areas in Glacial Lake Albany: Albany Pine Bush, Saratoga West, Saratoga Sandplains, and Queensbury (Fig. 1). From 2005 to 2007, our field crews sampled 41 patches varying in size from 0.24 to 10.3 ha. This accounted for about a third of the available KBB habitat area in Glacial Lake Albany at the time, and about 80 % of the area known to have been continuously occupied from 2001 to 2005 (New York State Department of Environmental Conservation, unpublished data).

Measurable Indicators and Rating Scheme

We followed a previously established protocol for determining indicators of conservation success (Parrish et al. 2003; Gordon et al. 2006). An indicator is defined as providing quantifiable, precise, relevant, and sensitive information about a key ecological attribute. The key attribute is a characteristic or requirement of a conservation target (species, community, ecosystem) that if degraded or removed would jeopardize the target's viability. The New York KBB recovery team identified habitat potential as one of several key attributes for the conservation target (i.e., the KBB in Glacial Lake Albany), and selected two indicators at the scale of recovery areas: total hectares of high-potential habitat and total number of lupine stems. "High-potential habitat" in turn was quantified by a set of indicators representing KBB habitat requirements measured at the scale of individual patches; these included lupine density, nectar diversity (species richness, density, evenness), and vegetation structure (dense-foliage grass cover, canopy cover, shade heterogeneity) as defined in Table 1. The goal with indicators is not to measure everything that may affect the conservation target, but instead focus on what is most important to the target's persistence (Lindenmayer 1999).

The range of indicator measurements can be categorized into a rating scheme useful for supporting management decisions. Here we followed a four-category scheme (poor, fair, good, very good) for synthesizing many indicators and communicating complex scientific information to decision makers (Vora 1997; Parkes et al. 2003; Parrish et al. 2003; Gordon et al. 2006). The top two categories (i.e., good and very good) represent the range of ecological conditions with high habitat potential, analogous to the acceptable range of variation (sensu Landres et al. 1999). The very good level represents an ideal condition that suggests no further corrective action is needed, whereas the good rating suggests active intervention may still be necessary to maintain the desirable condition (Parrish et al. 2003). The bottom two categories (i.e., fair and poor) suggest the target is not "conserved," or in our case, that habitat has low potential; a poor rating suggests greater urgency of conservation action than a fair rating. This four-category scheme aids prioritization by conveying more habitat information and accounting for more variation among sites than just a high- versus low-potential rating.

Despite limitations of categorical rating systems and combining individual metrics into a general index (Game et al. 2013; Hallett 2014), we chose to follow this method due to its ease of use and acceptance by managers. Indeed, the four-part rating system is now integrated with the widely adopted conservation-planning software Miradi™ (www.miradi.org). We viewed our approach as an interim step to understanding the mechanisms that define habitat quality. In the future, we expect to replace value judgments (see next section) with evidence to justify these selections, based on relational analysis with local KBB population estimates.

Establishing Rating Categories

Indicator rating categories were directly linked to specific viability criteria stated in the KBB state and federal recovery plans wherever possible (NYSDEC 1998; USFWS 2003), and supplemented by more recent research (e.g., Fuller 2008) and expert opinion. At the scale of recovery areas, total lupine was determined by summing point estimates and 95 % confidence limits of patch-level stem numbers across all patches sampled within a recovery area. Ratings were based on a range of carrying capacities (eggs per stem) needed to maintain a minimum viable KBB population for nine subpopulations (Fuller 2008, Table 4.6, p. 119). Categories included <575,000 (Poor), 575,000–775,000 (Fair), 775,000–1,150,000 (Good), and >1,150,000 (Very Good) total stems estimated in the recovery area.

The total hectares of high-potential habitat for each recovery area was based on summing the area of study sites that rated good or very good for the combined patch-scale indicators (combined using scorecard analysis, see "Habitat Potential Grading and Supporting Analyses" section). In addition, habitat patches were only included in the sum if they (1) were more than 0.25 ha in size (USFWS 2003, p. G-84)), (2) occurred within a subpopulation that was at least 5 ha in total size (based on NYSDEC 1998), and (3) occurred within 1 km of other subpopulations (the typical maximum KBB dispersal distance; Knutson et al. 1999; USFWS 2003). The good–very good boundary was set to the minimum recommended amount (258 ha) to support a large viable KBB population (USFWS 2003, p. F-67). Without further information to establish ecologically based criteria, this value was halved to create the good-fair cutoff, and halved again to create the fair-poor cutoff, yielding the following categories: <65 (Poor), 65–129 (Fair), 130–257 (Good), and ≥258 (Very Good) hectares of high-potential habitat in the recovery area.

At the patch scale (Table 1), lupine stem density ratings were established by dividing the above category boundaries for total lupine by the respective cutoffs for total high-potential habitat. Because the resulting good–very good cutoff (~4,450 stems/ha) is less than the resulting poor–fair cutoff (~8,900 stems/ha), we switched these values in setting the category boundaries (see Table 1). The ratings for nectar richness follow recommendations within the federal recovery plan (USFWS 2003) and more than two decades of restoration experience and field observations of KBB nectaring behavior. We set a lower requirement in spring richness because fewer KBB nectar sources flower during that time, and typically fewer KBB adults are recruited in the spring brood than summer brood (unpublished data). The nectar mean frequency and evenness (modified Simpson's index) categories were subjectively defined by quartiles following the simple assumption that more nectar abundance and evenness is better for butterflies. Greater abundance and evenness makes it more likely that at least some nectar species are available if unusual or catastrophic events eliminate other species, as recognized by King (2003).

Regarding habitat structure, native grass cover is represented by three dense-foliage species (*Andropogon gerardii*, *Schizachyrium scoparium*, *Sorghastrum nutans*) historically associated with pine barrens and believed important for KBB nocturnal roosting and egg laying (Smallidge et al. 1996; Benjamins 2003; USFWS 2003; Forrester et al. 2005). The rating levels correspond to our working hypothesis that too much grass cover (>70 %) crowds out food plants and makes them less apparent and too little (≤20 %) is insufficient for roosting sites (Table 1; Benjamins 2003; USFWS 2003). The desired cover range (5–30 %) for woody overhead canopy (>2 m) corresponds with open to partial canopy cover amounts reported to benefit KBB oviposition and larval growth (Grundel et al.

1998a, b; Lane and Andow 2003). For shade heterogeneity, we assumed that gap and shade intervals influence KBB microhabitat selection (e.g., for oviposition), activity mode (perching vs. flying), thermal regulation, and larval host plant quality and survival (Smallidge et al. 1996; Grundel et al. 1998a, b; Benjamins 2003; Lane and Andow 2003; Forrester et al. 2005; Pickens and Root 2008; Pavlovic and Grundel 2009). Heterogeneity is considered ideal (very good status) when shade is contributed by both trees and shrubs, and over-shading is assumed when more than 60 % of transects have >30 % shade cover density (inferred from Grundel et al. 1998a, b; Lane and Andow 2003).

Data Collection Protocol

Field methods employed to gather indicator data are detailed in internal reports that are available upon request, and portions of the methods along with alternatives have been rigorously evaluated (Bried 2009, 2013). Restricted random sampling (Elzinga et al. 1998) was used at smaller patches ($\leq$1.0 ha; $n = 17$) or for patches with narrow and elongated shape, such as powerline corridors. With this method, a baseline was established along the longest axis of the patch and divided into 10 m intervals. Sampling units (2×30 m^2 transects) were placed perpendicular to the baseline and evenly spread across intervals with no more than one transect per interval to ensure sample coverage throughout the patch. Within each interval, the location of a transect was randomly selected from 0 to 8 m distance, keeping a 2-m buffer to accommodate nectar plots (see below). Enough transects were established initially to cover 5 % of patch area, with a 15 transect minimum. Any extra transects, as determined from sample size predictions (see Bried 2013), were added at random to the intervals.

Simple random sampling was used for the larger habitat patches (>1.0 ha; $n = 24$). Sampling units were established as 2×30 m^2 cells of a grid created within the boundaries of each patch using ArcGIS v9.1. Enough cells were randomly selected using the Hawth's Analysis Tools extension for ArcGIS to cover 10 % of patch area. One-quarter or one-half of this selection was sampled initially (depending on patch size) with a 15-cell minimum. The remainder of the sample was used as backup in case initial units had to be omitted (e.g., >25 % outside patch boundary, landed along hiking trail) or sample size predictions called for more units.

For both designs, lupine stems were counted in 0.25 m^2 quadrats placed every meter along transects in dense lupine populations, or by searching a 2-m width along the transect length in sparse lupine populations (Bried 2013). Nectar and habitat structure information was collected on a randomly selected subset of transects that covered at least 2.5 % of the patch area. Presence of nectar species was tallied in equally spaced 4.0 m^2 quadrats (i.e., up to 8 quadrats spaced 2 m apart). For the nectar frequency and evenness indicators, we used frequency of presence across quadrats as a proxy for relative abundance. Structural attributes were sampled by point-intercept along every meter of transect, using a periscope densitometer to capture intercepts of woody canopy (>2 m height). Woody vegetation was classified as shrubs versus trees based on the particular growth form (multiple vs. single stem) observed at each point. All data used for this paper were collected in May, June, and July during 2005–2007.

Habitat Potential Grading and Supporting Analyses

We synthesized the Table 1 measurements into patch-scale estimates of habitat potential using The Nature Conservancy's Conservation Action Planning Workbook, Version 6b. This spreadsheet-based scorecard tool (sensu Stem et al. 2005) follows the Open Standards for Conservation (Schwartz et al. 2012) and is now integrated with the Miradi™ conservation-planning software (www.miradi.org). We assigned a rating category to each indicator measurement following the scheme in Table 1. Categories were then assigned the following numerical scores: Poor = 1.0, Fair = 2.5, Good = 3.5, Very Good = 4. The scores for each indicator were then averaged into an overall habitat grade for each patch, using the ranges of 1.0–1.745 (Poor), 1.75–2.995 (Fair), 3.0–3.745 (Good), and 3.75–4.0 (Very Good). An overall score of Good is not attainable if any habitat indicator rates Fair or Poor. Only those patches that received an overall Good or Very Good grade were considered as contributing to high-potential habitat availability.

As a form of sensitivity analysis, we repeated the assessment under four indicator scenarios: observed, best-case, worst-case, and exclusion. Observed ratings were based on the measurements for each indicator as defined in Table 1. Best-case uses the upper 95 % confidence limits for lupine density and nectar frequency, and the highest habitat potential ranking (among point estimate and confidence limits) for grass and overstory cover. Worst-case uses the lower 95 % confidence limits for lupine density and nectar frequency, and the lowest habitat potential ranking (among point estimate and confidence limits) for grass and overstory cover. The grass and overstory indicators were treated differently because their rating schemes are not unidirectional, i.e., more grass or overstory cover is not necessarily better (see Table 1), thus the upper confidence limit may put the indicator in a worse rating category than the point estimate. Exclusion ratings follow the removal of highly correlated indicators, as determined by ordination vector overlays (explained below) and pairwise rank associations (Spearman's test). Indicators were removed based on their measurement relatedness (e.g., overstory embedded within shade heterogeneity) and to minimize correlations among remaining indicators.

We evaluated the influence of single indicators to overall habitat potential ratings by removing each indicator one at a time ("leave-one-out" strategy). Shifts in rating category and the degree of change (1, 2, or 3 rating levels) were tallied across patches for each removal; direction of change is specific to the case study and thus not reported. We also ran principal components analysis to ordinate the patches and their habitat potential ratings in a multidimensional space defined by the indicators. If habitat potential corresponds to a certain indicator, then one could expect shorter Euclidean distances among patches (i.e., stronger grouping) in the same rating category. We overlaid the diagram with vectors depicting indicator relatedness (vector direction) and the relative influence of each indicator (vector length) in forming the ordination space. All indicators were adjusted to standard deviates (z-scores) prior to ordination.

Results

Across the Glacial Lake Albany recovery unit, lupine density showed good or very good ratings in 29 (71 %) habitat patches (Table 2). All except one of the remaining patches rated poor. Most of the poor lupine ratings (91 %) came from the Albany Pine Bush recovery area, in semi-open scrub oak barrens habitat or recently planted fields created after removing exotic trees. This contrasts with the Saratoga Sandplains where no patches rated poor (Table 2); however, one patch (N1) was borderline. Most of the 41 patches (76–98 %) showed good to very good ratings for nectar diversity indicators, whereas more than half lacked (poor rating) overstory cover and shade heterogeneity (Table 2). Saratoga Sandplains had proportionately fewer patches with poor ratings for grass cover, overstory cover, and shade heterogeneity (i.e., the habitat structure indicators), and proportionately more patches with very good scores for grass and overstory, compared to the Albany Pine Bush. However, the Albany Pine Bush had proportionately fewer fair-rated patches for all three structure indicators, and proportionately more good-rated patches for grass and overstory, compared to Saratoga Sandplains (Table 2).

Combining the habitat indicators through scorecard analysis showed that fewer than 20 % of all 41 habitat patches scored as good or very good, over a third (37 %) as poor, and the remainder (44 %) as fair ('Observed' column in Table 3). A proportionately greater number of patches were rated as poor/fair at the Albany Pine Bush than at the Saratoga Sandplains (G-test of independence, $G = 6.73$, $P = 0.0095$); Saratoga West and Queensbury contained too few sampled patches (<10) to analyze in this manner. Most of these poor/fair patches at the Albany Pine Bush are either shrubby pine barrens with sparse lupine density or recently created and planted fields with limited structural heterogeneity. All patches in Queensbury and Saratoga West were rated as poor/fair (Table 3).

Under the worst-case scenario all patches received poor or fair scores, whereas under the best-case scenario the number of good or very good ratings improved to 29 % of total patches (Table 3). In the best-case scenario, the number of poor/fair patches did not change at the Albany Pine Bush and decreased at Saratoga Sandplains, increasing the degree of difference between these recovery areas ($G = 12.30$, $P = 0.0005$). The exclusion scenario led to good or very good ratings in 24 % of total patches (Table 3). Ordination vector overlays suggested redundancy between overstory cover and shade heterogeneity, and among the nectar richness and frequency measurements (Fig. 2). These patterns were confirmed by strong rank correlation between overstory and shade ($n = 41$ patches, Spearman's $r = 0.90$), and between spring and summer nectar richness ($r = 0.66$); rank correlations among the five remaining indicators did not exceed 0.34. Given these correlation patterns, and that the measurement of overstory is included within the measurement of shade heterogeneity and richness is related to (and less rigorous than) frequency and evenness, we removed overstory and both richness indicators for the exclusion scenario.

In the leave-one-out analysis, removal of lupine and each structure indicator caused the most shifts in habitat potential ratings, compared to almost no changes following removal of nectar indicators (Table 4). The indicator rating changed by two levels at six patches, and one patch (A3) improved from poor to very good when ignoring lupine density (Table 4). This suggests lupine and structure indicators were more important than nectar indicators in creating variation among habitat potential ratings. Furthermore, in the ordination space patches did not form obvious clusters with regard to habitat potential ratings (Fig. 2), suggesting that no single indicator drove the rating estimates across all patches.

Only one subpopulation (N in Saratoga Sandplains) exceeded 5 ha of high-potential habitat, and only when the best-case scenario was assumed (Table 3). Therefore, total hectares of high-potential habitat scored well within the poor category (<65 ha) for each recovery area. We present this information purely for demonstration purposes because the total area sampled in each recovery area was below 65 ha to begin with (Table 3). In terms of estimated total lupine stems, both the Albany Pine Bush and Saratoga Sandplains exceeded the fair/good boundary (775,000

Table 2 Indicator measurements (as defined in Table 1) for each Karner blue butterfly habitat patch surveyed in the Glacial Lake Albany federal recovery unit during 2005–2007, along with distribution of the 41 patches across the rating categories for each indicator (at the bottom)

Recovery area	Subpop	Patch	Lupine	Spr. rich.	Sum. rich.	Nect. frequency	Nect. evenness	Grass	Overstory	Shade
Albany Pine Bush	A	1	94,454	4	5	83.6	78.9	46	24	31.1
		2	41,301	6	6	85.3	72.0	3	3	0
		3	2,893	5	4	91.1	77.1	38	15	22.6
	B	1	40,6185	1	4	10.0	89.2	24	26	60.0
		2	360,000	3	6	13.0	77.1	13	0.0	0
		3	461	6	6	81.3	72.1	17	4	0
		4	2,368	6	7	46.0	87.9	34	6	0
		5	35	4	3	62.7	84.6	1	10	7.0
		6	127	7	7	90.8	57.8	5	14	37.7
	C	1	13,156	4	4	96.1	70.9	65	0	3.6
		2	1,480	4	3	97.1	57.3	25	0	0
		3	159	5	5	46.2	82.5	3	9	9.0
	D	1	31,857	5	5	60.3	70.2	28	1	0
	E	1	65,015	4	2	75.4	65.3	52	0	0
		2	87,608	3	2	57.3	84.4	69	0	0
		3	63,357	6	4	71.0	80.7	59	0	0
		4	2,445	4	2	89.3	77.5	31	0	0
	F	1	1,398	4	5	32.5	90.7	8	6	7.3
	G	1	607	5	4	62.5	68.4	2	3	4.0
	H	1	378,667	3	3	59.2	67.0	38	25	40.0
Queensbury	I	1	31,313	5	2	33.3	71.0	35	1	26.7
	J	1	14,222	4	4	49.2	74.1	4	11	1.0
Saratoga Sandplains	K	1	56,584	4	2	92.5	38.7	18	53	93.3
	L	1	7,507	5	6	92.3	77.3	15	14	16.7
		2	87,884	3	3	60.8	58.0	60	27	33.3
	M	1	81,041	4	3	97.4	81.6	0	41	66.7
	N	1	4,993	5	6	83.6	78.9	10	0	0
		2	47,524	5	6	91.7	66.5	13	35	47.6
		3	63,604	6	6	92.9	63.9	33	38	54.1
		4	66,476	3	3	96.7	64.9	29	12	13.3
		5	81,147	3	2	75.0	64.3	83	10	10.0
	O	1	44,302	4	4	98.8	80.9	43	4	0
	P	1	112,862	4	6	84.1	71.0	63	12	22.5
Saratoga West	Q	1	42,067	1	2	8.8	37.1	40	43	65.0
	R	1	7,386	4	3	50.8	74.1	65	0	0
		2	7,635	3	2	53.3	79.2	68	0	0
		3	6,062	3	3	41.2	70.9	60	0	0
		4	6,373	2	1	20.4	97.9	70	0	0
		5	2,477	3	3	53.9	79.1	55	0	0
	S	1	16,026	3	3	65.0	83.3	14	31	46.7
	T	1	30,239	3	2	82.8	65.7	49	4	6.7
No. patches in Poor category			11	0	0	4	0	6	21	21
No. patches in Fair category			1	2	1	6	2	10	5	10
No. patches in Good category			5	12	26	12	20	15	4	7
No. patches in V. Good category			24	27	14	19	19	10	11	3

'Subpop' = subpopulation, patches less than 200 m apart from each other (i.e., within typical KBB inter-patch movement distance) were assigned to the same subpopulation

Table 3 Rating of habitat potential for Karner blue butterfly habitat patches monitored in the Glacial Lake Albany federal recovery unit during 2005–2007

Recovery area	Subpop	Patch	Patch size (ha)	Restoration activity[a]	Estimated habitat potential[b]			
					Observed	Worst-case	Best-case	Exclusion
Albany Pine Bush	A	1	0.97	None	V. Good	Fair	V. Good	V. Good
		2	1.38	O	Poor	Poor	Poor	Poor
		3	3.72	L	Poor	Poor	Fair	Poor
	B	1	0.45	None	Fair	Fair	Fair	Fair
		2	0.57	P	Poor	Poor	Fair	Fair
		3	1.38	L, O	Poor	Poor	Poor	Poor
		4	2.23	L	Poor	Poor	Fair	Poor
		5	10.28	M&B, Pl	Poor	Poor	Poor	Poor
		6	6.47	M&B, Pl	Poor	Poor	Poor	Poor
	C	1	6.60	L	Fair	Poor	Fair	Fair
		2	2.06	L	Poor	Poor	Poor	Poor
		3	9.59	M&B, Pl	Poor	Poor	Poor	Poor
	D	1	0.73	O	Fair	Poor	Fair	Fair
	E	1	0.49	O	Fair	Poor	Fair	Fair
		2	0.28	O	Fair	Poor	Fair	Fair
		3	0.93	O	Fair	Poor	Fair	Fair
		4	1.01	O	Poor	Poor	Poor	Poor
	F	1	5.06	O	Poor	Poor	Poor	Poor
	G	1	3.04	O	Poor	Poor	Poor	Poor
	H	1	0.89	O	Good	Fair	V. Good	V. Good
Queensbury	I	1	0.24	None	Fair	Fair	Fair	Good
	J	1	0.36	None	Fair	Poor	Fair	Poor
Saratoga Sandplains	K	1	0.36	None	Poor	Poor	Fair	Fair
	L	1	1.38	C, P	Good	Poor	Good	Fair
		2	0.32	Pn	Good	Fair	Good	Good
	M	1	0.28	C, P	Fair	Poor	Fair	Fair
	N	1	1.38	O, P	Poor	Poor	Poor	Fair
		2	1.98	C, P	Fair	Fair	V. Good	V. Good
		3	2.63	C, P	V. Good	Fair	V. Good	V. Good
		4	0.85	Pn	Good	Fair	Good	V. Good
		5	0.24	Pln	Good	Poor	Good	Fair
	O	1	2.63	C, P	Fair	Poor	V. Good	Fair
	P	1	2.43	None	V. Good	Fair	V. Good	V. Good
Saratoga West	Q	1	1.17	C, Pln	Fair	Fair	Fair	Fair
	R	1	0.28	M, Pn	Fair	Poor	Fair	Fair
		2	0.45	M, Pn	Fair	Poor	Fair	Fair
		3	2.87	M, Pn	Fair	Poor	Fair	Fair
		4	4.21	M, Pn	Fair	Poor	Fair	Fair
		5	1.13	M, Pn	Poor	Poor	Poor	Poor
	S	1	0.89	M, Pn	Fair	Fair	V. Good	V. Good
	T	1	0.61	M, Pn	Fair	Fair	Good	V. Good

[a] Habitat patch B2 was converted from a paved parking lot and only about half the area of patch H1 is restored old field

[b] Habitat potential ratings were mathematically pooled (via the Conservation Action Planning Workbook, Version 6b) from the eight indicators described in Table 1. 'Observed' ratings are based on the measurements for each indicator found in Table 2. 'Worst-case' substitutes the lower 95 % confidence limits for lupine density and nectar frequency, and the lowest habitat potential ranking (among point estimate and confidence limits) for grass and overstory cover. 'Best-case' substitutes the upper 95 % confidence limits for lupine density and nectar frequency, and the highest habitat potential ranking (among point estimate and confidence limits) for grass and overstory cover. 'Exclusion' ratings follow the removal of three redundant indicators (overstory cover, spring and summer nectar richness) as explained in the text

O old field conversion, *L* black locust removal, *P* planted lupine, nectar, and native grasses (*Pn* planted nectar only, *Pl* planted lupine only, *Pln* planted lupine and nectar only), *M* mowing, *M&B* mowing and prescribed fire, *C* forest canopy thinning, *none* no active management

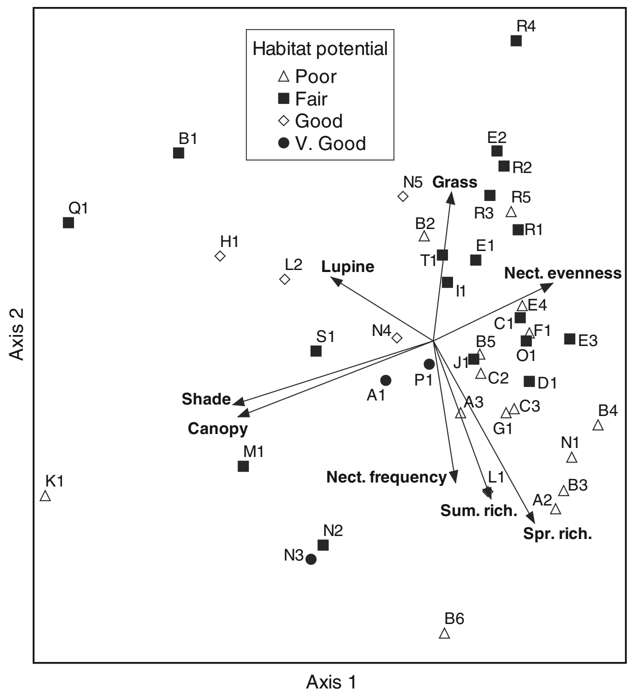

Fig. 2 Principal components analysis of the indicator results in Table 2, with patches categorized according to observed estimates of habitat potential in Table 3. The first *two axes* cumulatively extracted 59.5 % of variance in the relativized data matrix, with good correlation (75.3 %) between distances in the ordination diagram and distances in the matrix. The *letter–number label* for each point corresponds to subpopulation and habitat patch codes listed in Tables 2 and 3. *Vector lengths* represent the size of each indicator's contribution to forming the ordination space, and *vector directions* suggest degree of relatedness between indicators

stems) based on the average estimate, and the good/very good boundary (1,150,000 stems) based on the maximum estimate (Fig. 3). However, the minimum estimates for these recovery areas rated as poor. The whole indicator range for Saratoga West was well below the poor/fair boundary (Fig. 3), and the two patches sampled in Queensbury (not shown in Fig. 3) contained an estimated total of about 12,600 stems within a range of 3,950–21,300 stems.

Discussion

We have outlined and implemented a framework to indirectly assess the potential for endangered insect recovery through habitat monitoring and restoration objectives. The framework provides a transparent process for progressively and systematically defining habitat potential, refining its measurement, and addressing habitat restoration priorities as time and resources allow. This is particularly beneficial for insects because accurate population estimates are often difficult to attain and track over time, making recovery difficult to assess. The concepts and elements are not new (see for example Vora 1997; Parkes et al. 2003), but applications to species management are scarce and

Table 4 Shifts in habitat potential rating and the degree of change (1, 2, or 3 rating levels) tallied across patches following removal of each habitat indicator one at a time

Excluded indicator	No. patches		
	One level	Two levels	Three levels
Lupine density	7	3	1
Spring nectar richness	0	0	0
Summer nectar richness	0	0	0
Nectar frequency	1	0	0
Nectar evenness	0	0	0
Grass cover	11	2	0
Overstory cover	10	1	0
Shade heterogeneity	5	0	0

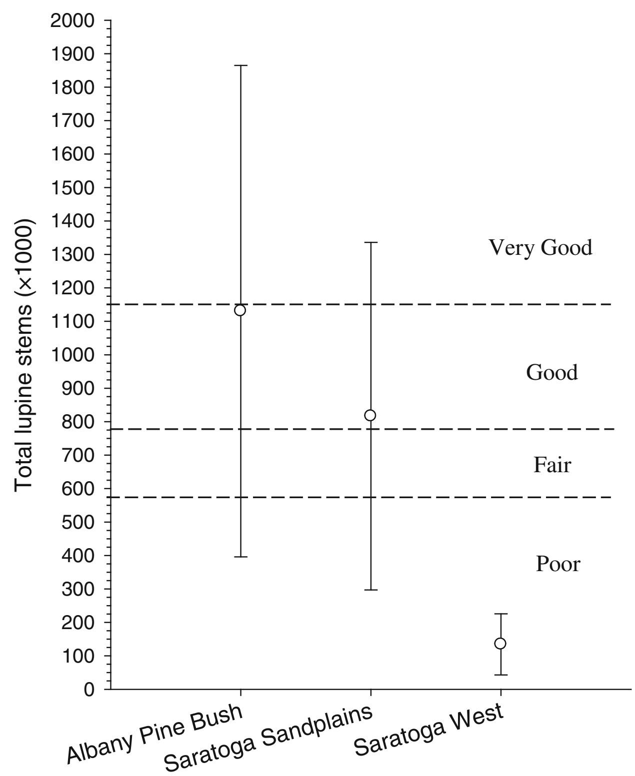

Fig. 3 Estimated lupine abundance at the scale of recovery areas, excluding Queensbury where only two patches were sampled. The point estimates come from extrapolating lupine density out to patch area and then summing the stem totals across the patches within each recovery area. The minimum and maximum values follow this same approach but correspond to lower and upper 95 % confidence limits on lupine density. *Dashed lines* indicate the rating category boundaries explained in the text

published examples for endangered insects are non-existent. Clearly the application to KBB recovery is a work in progress and the habitat assumptions require relational analysis with local KBB population estimates for validation. However, it is important to recognize that despite having limited knowledge in many aspects of endangered insect recovery, the framework can be consistently applied so that information is useful in prioritizing recovery areas and sites for management.

Application to KBB Recovery

Our assessment made it possible to identify which recovery areas are worth investing conservation resources in and where site-specific restoration work is needed. The Albany Pine Bush and Saratoga Sandplains emerged as priority recovery areas, because they have the most habitat potential and infrastructure to support restoration (i.e., enough protected land base, strong financial and sociopolitical backing). Saratoga Sandplains tended to have higher habitat potential than Albany Pine Bush and may require less effort to meet restoration goals. This may simply have resulted from differences in restoration starting points or vegetation community types (Bried 2013) than from greater success of restoration efforts at Saratoga Sandplains. Within these recovery areas, the assessment has helped managers prioritize sites for habitat improvement and remediation.

Reducing woody canopy cover, both to increase patch size and promote better shade heterogeneity (from trees and shrubs), was identified as a priority site-level action, consistent with previous findings that limited and variable canopy structure promotes KBB and lupine survival (Grundel et al. 1998a, b; Pickens and Root 2008; Pavlovic and Grundel 2009). Furthermore, although estimated lupine abundance may be sufficiently high across select recovery areas (Fig. 3), low lupine density at the patch-scale, particularly in pine barrens habitat, suggests heavier lupine inter-seeding, lighter rates of grass planting (to mitigate outcompeting lupine seedlings), and more intensive thinning of shrubs and trees as needed on a site-by-site basis. Table 2 contains additional site-specific information useful to guide management actions across Glacial Lake Albany.

Given the background variation among recovery areas and the many types of management actions employed (listed in Table 3), it is difficult to directly determine the contribution of restoration to estimated habitat potential without before–after surveys. Nevertheless, our assessment still provides some evidence for the success of past restoration efforts. For example, the majority of habitat patches that achieved good or very good ratings received some form of management action. Similarly, in the only subpopulation close to having high habitat potential (subpopulation N in Saratoga Sandplains), all five patches experienced recent restoration activity. As we progress with continued restoration work, the current assessment will provide an important baseline for tracking changes within each of these sites, and potentially for evaluating management activities in the future as more patches are added to the monitoring program.

Our overarching goal in this paper was to demonstrate a framework for monitoring habitat potential and restoration progress in the context of endangered insect recovery. Specific results from the case study should be viewed and used with caution for at least two reasons. First, many potential habitat patches in Glacial Lake Albany (over 400 mapped currently) were not sampled, and extensive habitat creation and restoration planting has occurred since 2007, especially in the Albany Pine Bush and Saratoga Sandplains. We know from follow-up surveys at several patches that lupine populations are expanding, quite dramatically in some cases. For example, lupine density increased in one patch (G1) from about 600 stems/ha (poor rating based on Table 1) in 2007 to nearly 10,000 stems/ha (very good rating) in 2009. Second, the sampling scheme may have caused severe underestimates of lupine abundance in some cases, especially in shrub barrens at the Albany Pine Bush where lupine populations are diffuse and clustered (Bried 2013); indeed these patches comprised most of the poor ratings for lupine density (<4,450 stems/ha). The amount of high-potential habitat and lupine estimated in this study does not reflect the current habitat-based recovery status in Glacial Lake Albany.

Recovery plan goals and objectives are often biased low (Tear et al. 1993, 1995), and recent research suggests that some criteria in the KBB recovery plan are no exception (Fuller 2008). The state recovery team will be re-evaluating the recovery thresholds and habitat potential criteria by testing relationships between KBB abundance and the current habitat requirement indicators, along with other potentially important environmental factors that were not analyzed such as topography, snowpack, and patch size. To this end, the state recovery team has identified population-based indicators of recovery and is currently analyzing patterns in peak counts, population densities, and brood sizes with the intent of linking them to updated estimates of habitat potential. Ultimately the team will integrate population, habitat, and landscape (e.g., subpopulation connectivity) components into a robust evaluation of KBB conservation status.

Technical Considerations for Applying the Framework

Ordination and leave-one-out analyses suggested that some habitat requirement indicators were more influential than others, but that no single indicator drove the ratings across all patches. For example, failure for patches to rate as good was not consistently due to any particular indicator or subset of indicators, and not all indicators had to rate as poor for a patch to be considered poor. Thus, it appears that no single indicator can be used to estimate overall habitat potential, and managers should avoid generalizations about habitat suitability based on single indicators. This may be especially true across large and variegated recovery areas, and because organisms generally respond to multiple habitat factors.

The use of several metrics to provide multiple lines of evidence may compensate for weaknesses of individual metrics. Indeed, multimetric indices for integrative assessments of ecological integrity assume that different metrics provide unique information and therefore a variety of metrics should increase power and sensitivity of the assessment (Schoolmaster et al. 2012). However, too many metrics relative to sample size, or too many redundant metrics, will reduce power/sensitivity (Van Sickle 2010). Therefore, the general problem is to select a limited set of uncorrelated metrics from a potentially large number of candidates. We found clear redundancy among certain habitat indicators based on the ordination vector overlays and pairwise rank correlations. Our analysis suggests that removal of correlated indicators can improve the habitat potential ratings (compare the Observed and Exclusion columns in Table 3).

The concepts and descriptive categories (poor, fair, good, very good) are well justified (Parrish et al. 2003) and found in other ecological condition assessments (Vora 1997; Parkes et al. 2003; Hallett 2014). However the system requires conscientious usage not just in the selection of indicators and category boundaries but also for potential bias or skew toward certain ranking categories (Hallett 2014). Category weighting can be applied when combining attributes of size (population or area), condition (e.g., habitat potential), and landscape context (e.g., connectivity, land cover). For example, in determining the viability of KBB across a recovery unit, assessors may decide to down-weight the habitat potential scores (e.g., by a factor of 0.5) to better emphasize KBB brood size and subpopulation connectivity. Weighting each indicator is not necessary because they presumably link to the same ecological attribute (habitat potential in this case). However, managers should consider the measurement location of each indicator within the rating category, i.e., is it near the center or at the boundaries? As our example illustrates, two sites may score in the same category yet at one site the indicator measurement is centered and at the other lies near a boundary. Game et al. (2013) provide a more detailed discussion of the caveats and limitations to the scorecard system.

Conservation and Recovery Implications

Recovery of a species to the point of downlisting or delisting is likely to be a long process. As such, establishing short- and long-term goals is important to guide conservation (Tear et al. 2005). Here we focused on shorter-term goals associated with habitat potential, which is closely tied to restoration activities and serves as a proximate indicator for assessing whether species recovery will follow the desired trajectory. The proposed framework converts quantitative habitat indicators to descriptive habitat potential categories ranging from unacceptable (i.e., poor rating) to ideal (i.e., very good rating). These ratings provide benchmarks for setting transparent objectives for habitat restoration and recognize that restoration should be viewed more as a continuum than a discrete or static condition of restored versus not restored, similar to the recovery process (Scott et al. 2005).

The habitat potential grading scheme is derived from recovery plans and the latest information on species biology and includes many working hypotheses (sensu Tear et al. 2005) regarding key ecological attributes, indicators, and rating levels. Establishing benchmarks to evaluate recovery progress has been missing in many endangered species recovery efforts, not just butterflies (Gerber and Hatch 2002; Lundquist et al. 2002). By selecting measurable objectives via this process, conservation planners can define and track progress more explicitly, a critical task often neglected in restoration ecology (Ruiz-Jaen and Aide 2005; Wortley et al. 2013). Importantly, this approach can be extended to other aspects of recovery, such as using target species population size, rate of change, and migration among patches as indicators of the ultimate recovery attribute: population status. Ideally, the framework would be complemented by other important aspects of endangered species recovery, such as multiple threats assessment and information quality control (Bried 2009; Murphy and Weiland 2011; Bernazzani et al. 2012; Darst et al. 2013).

The proposed framework offers a convenient method of communicating complex scientific information to conservation planners and resource managers. It also helps restoration ecologists and practitioners incorporate their best professional judgment and working hypotheses in the recovery planning process. Expert opinion and manager experience have an important place in conservation, and decisions or courses of action that evolve from imperfect knowledge are often effective and necessary (Tear et al. 2005; Bried and Mazzacano 2010; Murphy and Weiland 2011; Litvaitis et al. 2013). Such thinking is applicable to many species that require active management to persist, and extends beyond U.S. endangered species recovery. With so many species in steep decline, conservation planners should be proactive and not delay action in wait of improved knowledge (Grantham et al. 2009). This could mean combining habitat assessments with monitoring extinction risk (Staples et al. 2005) to give a more comprehensive view of restoration progress and allow managers to take action before rather than during a serious population decline.

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Ethical standards All data collection and reporting comply with current US laws, including the Endangered Species Act.

Conflict of interest The authors declare that they have no conflict of interest.

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