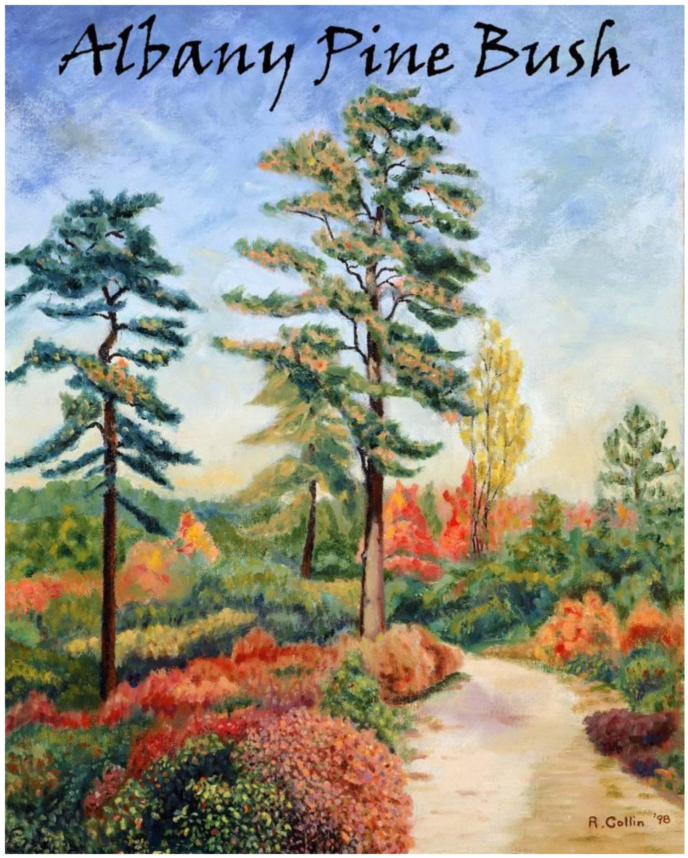
Appendix B.

Albany Pine Bush Pine Barrens Viability Assessment

Albany Pine Bush Pine Barrens Viability Assessment

Quantifiable indicators of pine barrens size and extent, fragmentation and edge effects, prescribed fire regime, and biotic patterns



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Albany Pine Bush Preserve Commission

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Acknowledgements

Participants of the November 2004 planning workshop (see p. 5) laid the foundation for this project. The workshop was organized by Stephanie Gifford and funded by the New York State Biodiversity Research Institute and the Northeast Fire Learning Network. Bob Collin did the artwork that graces the cover.

I. INTRODUCTION

Temperate grasslands, savannas, and shrublands form the most highly altered and least protected terrestrial biome (Hoekstra et al. 2005). In the northeastern U.S., shrublands dominated by pitch pine (*Pinus rigida*) and scrub oaks (*Quercus ilicifolia*, *Q. prinoides*) are in rapid decline and among the highest priorities for conservation (Noss et al. 1995, Neill 2007). A motley assortment of inland pine barrens are scattered from Pennsylvania to Maine on coarse sand deposits or bare rock outcrops (Seischab & Bernard 1991). Prime examples include the Pocono Plateau of Pennsylvania, Albany Pine Bush in New York, Montague Sand Plain in Massachusetts, Concord and Ossipee barrens of New Hampshire, and Waterboro barrens in Maine (Forman 1979, Latham et al. 1996, Motzkin et al. 1996, Finton 1998, Copenheaver et al. 2000). Many rare, threatened or endangered species depend on the region‟s shrublands and barrens (Dettmers 2003, Latham 2003, Wagner et al. 2003). The primary threat to these early successional communities is repression of frequent disturbances such as fire and land clearing for agriculture (Lorimer & White 2003).

In human dominated regions like the northeastern U.S., active management is often necessary to maintain natural systems and taxa dependent on disturbance (DeGraaf & Yamasaki 2003, Scott et al. 2005). The science and practice of simulating disturbance for restoring northeastern U.S. shrublands has greatly matured [see *Biological Conservation* 136(1) and *Forest Ecology and Management* 185(1-2)], but far less is known or documented about measuring the resulting trajectories. This is despite general recognition that monitoring assessments may link restoration activity to more effective conservation decisions (Stem et al. 2005, Nichols & Williams 2006, Lovett et al. 2007).

The Albany Pine Bush Preserve Commission is interested in creating a biological monitoring system scaled to its pine barrens landscape management. To do this, Commission scientists have proposed quantifiable indicators of pine barrens size and extent, fragmentation and edge effects, prescribed fire regime, and biodiversity patterns. Together these indicators offer a viability assessment framework for estimating ambient status and restoration progress in the globally rare pitch pine-scrub oak community type. The Albany Pine Bush is the focal site for the proposed assessment, but the general model and much of the specifics are applicable to other northeastern U.S. pine barrens and sand plains.

Planning & Conservation Target

The project was enabled by funding from the New York State Biodiversity Research Institute awarded to the Eastern New York Chapter of The Nature Conservancy, and by funding from the Northeast Fire Learning Network awarded to the Conservancy‟s Shawangunk Ridge Program. The North American Fire Learning Network is a cooperative project of The Nature Conservancy, U.S. Forest Service, and U.S.

Department of Interior, and tries to promote ecologically appropriate fuels reduction and restoration projects at local and national levels.

An initial planning workshop, "*Preparing a Prescription for Success: Managing and Monitoring Eastern Pitch Pine Barrens and Oak Forests*", was held 18–19th November 2004 at the Minnewaska Lodge in Gardiner, New York. The workshop was led and organized by Stephanie Gifford, former Director of Ecological Management for The Nature Conservancy‟s Eastern New York Chapter. Pine barrens experts and representatives from seven states participated. The goal was to develop prescriptions for managing and monitoring eight pine barrens and oak forest systems in New York:

- Inland pitch pine-scrub oak barrens (Albany Pine Bush)
- Coastal oak-heath forest (Long Island)
- Pitch pine-oak forest (Albany Pine Bush/Long Island)
- Pitch pine-oak-heath woodland (Long Island)
- Dwarf pine plains (Long Island)
- Pitch pine-oak-heath-rocky summit (Shawangunk Ridge)
- Dwarf pine ridge (Shawangunk Ridge)
- Chestnut oak forest (Shawangunk Ridge)

This report focuses on the Albany Pine Bush and therefore the inland pitch pine-scrub oak barrens system. „Pitch pine-oak forest‟, which generally contains several tree oak species, is a lesser priority in the Albany Pine Bush and was not dealt with explicitly in the current assessment. The pitch pine-scrub oak barrens system consists of three successional variant communities, sometimes called "embedded communities":

Pitch pine-scrub oak barrens

Pitch pine-scrub oak thicket

Pitch pine-scrub oak forest

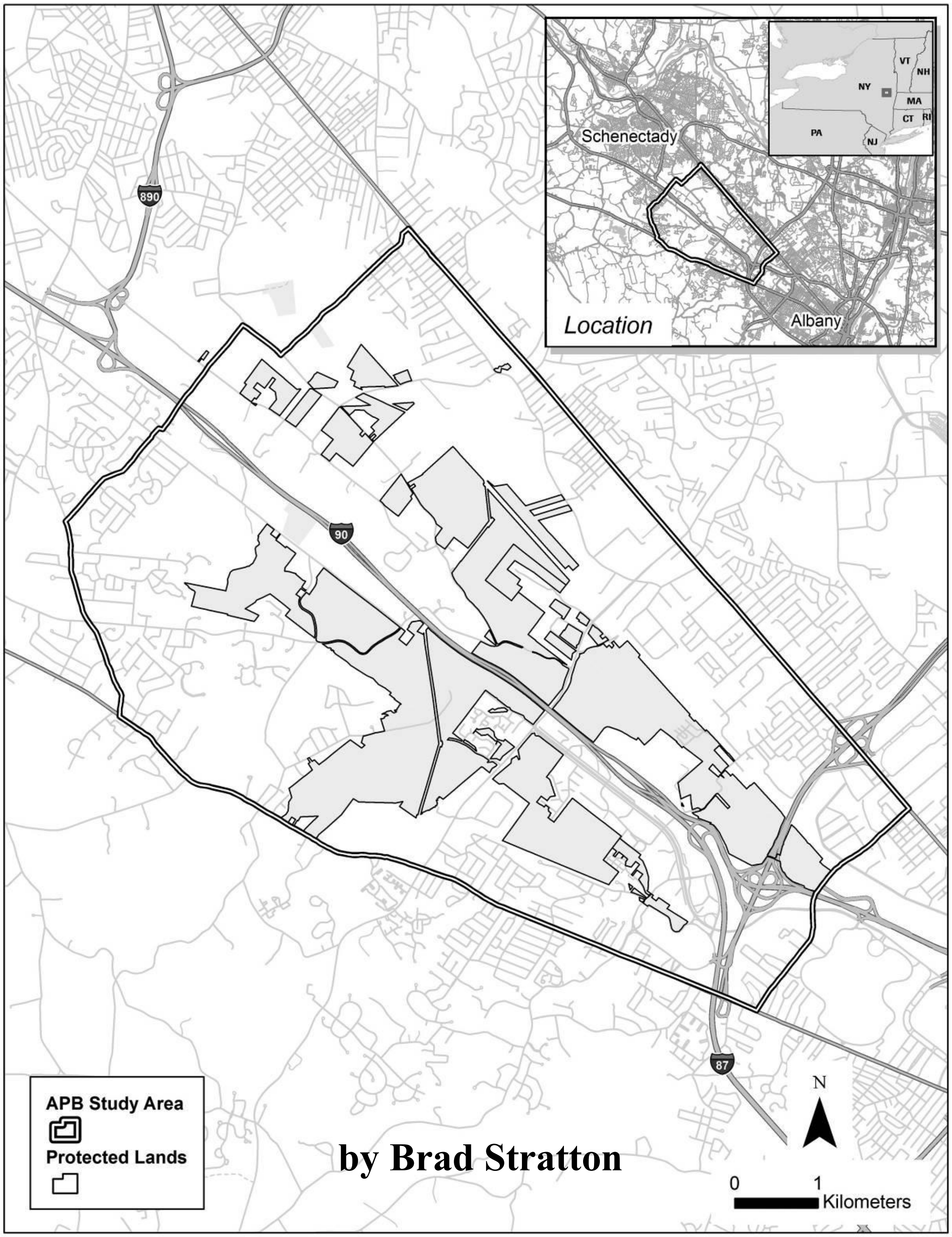
The term "pine barrens" commonly refers to these and other early successional or shrubland-type communities maintained by periodic fires and growing in well-drained, sandy soils. In Albany Pine Bush barrens the shrub layer is dominated by scrub oaks (*Quercus ilicifolia*, *Q. prinoides*) ranging from about 30–60% cover, whereas in thickets the scrub oak is taller and more dense (often 80–100% cover). Pitch pine (*Pinus rigida*) dominates the Albany Pine Bush forest variant (>60% cover) and tops scattered to thicket-forming scrub oak, having less cover (usually 20–60%) in barrens and thickets. The lower shrub or sub-shrub layer includes dwarf willows (*Salix humilis*, *S. tristis*), sweet fern (*Comptonia peregrina*), blueberries (*Vaccinium angustifolium, V. pallidum*), black huckleberry (*Gaylussacia baccata*), and sand cherry (*Prunus pumila*). The herb layer, often most prominent in prairie openings, includes big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indian grass (*Sorghastrum nutans*) plus characteristic forbs like bush clovers (*Lespedeza capitata*, *L. hirta*, *L. procumbens*, *L. stuevii*), goat‟s-rue (*Tephrosia virginiana*), and wild lupine (*Lupinus perennis*). Together the variant communities support many rare and relatively exclusive Lepidoptera, herpetofauna, and avifauna, such as the Karner blue butterfly, inland barrens buckmoth, prairie warbler, brown thrasher, eastern hognose snake, and eastern spadefoot toad (Edinger et al. 2002, NYNHP 2007).

Albany Pine Bush

The Albany Pine Bush Preserve (APB) in east-central New York State (see map below) protects one of the best and few examples of inland pitch pine-scrub oak communities worldwide. The APB pine barrens landscape historically spanned over 10,000 ha of sandy soils and was one of the largest inland areas of pine barrens vegetation in the glaciated northeastern U.S. (Gebauer et al. 1996, Barnes 2003). The present ~1,200 ha APB protected land base contains approximately 140 ha of scrub oak barrens and thicket regions and 255 ha of pitch pine-dominated forest. The larger APB study area (~5,000 ha), as delineated by historic sand dune topography, adds about 95 ha of barrens/thickets and 290 ha of pitch pine forest to the protected land base.

As a priority landscape project in the 2009 New York State Open Space Conservation Plan (http://www.dec.ny.gov/docs/ lands_forests_ pdf/osp2009.pdf) and the Sierra Club "America‟s Wild Legacy" conservation initiative, the APB is known for its diverse rare natural communities and species. The APB helps protect several rare plants (e.g., Bayard‟s Alder‟s-mouth Orchid, *Malaxis bayardii*) and like many shrublands with abundant scrub oak it harbors numerous rare animal species (Barnes 2003). It contains at least 44 state-designated Species of Greatest Conservation Need (New York Comprehensive Wildlife Conservation Strategy, http://www. wildlifeactionplans .org/new_york.html) among birds, amphibians, reptiles, and Lepidoptera. This total includes 28.2% of all the Species of Greatest Conservation Need in New York‟s 30,300 km^2 Upper Hudson Basin and 8.2% of the Species of Greatest Conservation Need statewide. The APB is also a U.S. designated metapopulation recovery area for the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis*), and contains prime examples of the rare and enigmatic Pine Barrens Vernal Pond (Bried & Edinger 2009).

Human settlement of the northeastern U.S. through the 20th century brought increased fire suppression, which is the leading threat to fire-dependant shrublands like pine barrens (Jordan et al. 2003, Lorimer & White 2003). Remnant pine barrens of the region are further stressed by invasive plants. Species causing the most economic strain and ecological damage in the APB are exotic black locust (*Robinia pseudoacacia*) and native aspens (*Populus grandidentata* and *P. tremuloides*). Black locust spreads vigorously from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root sprouting and in the APB has facilitated competitive exclusion of barrens-adapted vegetation by enriching soil nitrogen levels (Rice et al. 2004, Malcolm et al. 2008). Aspens take advantage of frost tolerance, fire suppression, and rapid clonal growth in spreading across large areas of the APB landscape (Milne 1985).

In response to these threats, the Albany Pine Bush Preserve Commission administers a comprehensive landscape restoration program (APBPC 2002). Commission staff, private contractors, and volunteers are working to create and sustain pitch pine-scrub oak communities using techniques like mowing and burning, whole-tree removal, planting native species, “clip-and-drip” herbiciding of black locust, and stripping bark layers (called “girdling”) to kill aspen (APBPC 2002, Barnes 2003). The focus of restoration *monitoring*, however, has so far been “fine-filter” and directed towards Karner blue butterfly habitat management (Bried 2009, Tear et al. unpublished data).

Coarse vs. Fine Filter Monitoring

The scale of conservation actions is typically thought of as being fine-filter (genes, populations, species) or coarse-filter (communities, ecosystems, landscapes) (Noss 1987); more recently a “meso-filter” strategy was proposed (Hunter 2005). One of the major shortcomings of coarse- or meso-filter conservation, compared to fine-filter, has been the lack of generalized and objective measures of success (Tear et al. 2005). Another issue centers on reconciling the need for fine-filter conservation while accommodating coarse-filter conservation. It is impossible to protect biodiversity species by species, but without special attention some at-risk taxa may “slip through the pores” of a coarse filter (Schwartz 1999). One of the dangers with fine-filter management, however, is that it may preclude consideration of and negatively affect non-target species (Krementz & Christie 1999).

Pine barrens conservation and management has historically taken more of a fine-filter approach, such as by using single species wildlife models (e.g., sharp-tailed grouse; Niemuth & Boyce 2004). The proposed pine barrens viability assessment detailed in this report is a coarse-filter approach targeting a community (pitch pine-scrub oak barrens, thicket, and forest), ecosystem (pine barrens biotic and abiotic elements), and landscape (Albany Pine Bush).

Assessment Framework

The Nature Conservancy has streamlined complex efforts to monitor changes in the health of conservation targets (species, communities, ecosystems) and to implement “corrective” management actions when needed (Groves et al. 2002, Parrish et al. 2003). Their approach forces clear objectives and detailed vision for what a conservation target should look like. Generally referred to as “conservation action planning” (CAP), The Nature Conservancy approach consists of three core components: key ecological attributes, indicators, and indicator ratings. Each category of information may be drawn from ecological models (conceptual or mathematical), best available science, expert consultations, local scientific data, and data from comparable targets in other locations.

The Nature Conservancy‟s CAP borrows from and builds upon existing ideas and language in ongoing efforts to broadly define ecosystem health (Brooks & Grant 1992, Woodley et al. 1993, Keddy & Drummond 1996, Vora 1997, Aplet & Keeton 1999, Landres et al. 1999, Schwartz 1999, Parkes et al. 2003). Examples of CAP applications are found in the literature (Moseley et al. 2004, Gordon et al. 2006, Tear et al. unpublished data) and via the open access ConserveOnline and Conservation by Design Gateway (http://conserveonline.org/workspaces/cbdgateway).

The proposed APB pine barrens viability assessment groups attributes into four categories (modified from Parrish et al. 2003): size and extent, fragmentation and edge effects, prescribed fire regime, and biodiversity and structure patterns. Altogether 18 attributes were selected, and are narrated in detail in the ensuing chapters (II-V):

	Size & Extent	**Fragmentation & Edge Effects**	**Rx Fire Regime**	**Biotic Patterns**
Key ecological attributes	Habitat amount Patch size Core area Suitable Karner blue butterfly habitat	Patchiness Patch isolation distance Perimeter/area ratio Edge effects from roads, trails, and residential	Refugia Individual fire size Return interval Seasonality	Cover of pitch pine and scrub oaks Floristic tolerance of human activity Invasive plant impact Reduction of priority invasive vegetation Characteristic rare Lepidoptera Shrubland birds

Key ecological attributes are characteristics of the target that if degraded (e.g., water quality) or removed (e.g., pollinator) would jeopardize the target‟s viability, or ability to persist over time. They are the essential currency for identifying and measuring the composition, structure, and function of the target. The point is not to worry about measuring everything but instead focus on what is key, or what is known or believed to influence the target‟s persistence the most (Parrish et al. 2003). This thinking is consistent with Lindenmayer (1999): „*…a key aspect of well-designed monitoring programs will be to ensure that they are well focused with a limited number of entities being studied*‟.

One or more quantifiable indicators are used to capture and estimate each attribute concept. Indicators need to be biological relevant, socially relevant (i.e., value is recognized by stakeholders), sensitive to anthropogenic stress, anticipatory (provide early warning), measurable, and cost-effective (Parrish et al. 2003). The relationship of attributes and indicators may be viewed as analogous to empirical modeling insofar as indicators being quantifiable variables used to estimate parameters (attributes) of interest. A target‟s indicator values will vary over time, and this change may be natural and consistent with long-term persistence of the target, or, fall outside the natural range because of human influence (e.g., fire suppression in fire maintained systems). A "conserved target" may therefore be defined as maintaining each attribute within their acceptable ranges of variation. Managing for an acceptable *range* of variation in each attribute is likely to be more beneficial than managing for a static pattern (Landres et al. 1999). The Nature Conservancy segments the range of indicator values (qualitative or quantitative) into four categories (i.e., poor, fair, good, and very good defined in Parrish et al. 2003, Gordon et al. 2006). The top two categories (good and very good) define the acceptable range of variation for each attribute and indicator; this rating scheme is also used to define the desired ecological conditions to guide management actions. The rationale for these thresholds is recorded for each indicator based on the best available science. In this report indicator ratings were set independent of what is currently considered feasible in the APB, and instead were based on what would be necessary for the target to sustain itself over time (e.g., area needed to support naturally occurring disturbance dynamics). Much of the rating system proposed here is based on the focal-species concept (Lambeck 1997), where the most area-demanding, dispersal-limited, and disturbance-sensitive species of concern set the benchmarks.

Since no single criterion (turnover rates, large area requirements, habitat specialization, etc) will capture the complexity of a managed ecosystem, indicators for monitoring and evaluation should be considered as a group (multi-metric analysis) rather than singly (Kremen 1992, Keddy & Drummond 1996). An attractive feature of the CAP framework is the recent development of tools for documenting the indicators and decision process, and quantifying target viability and threat levels. Data gathered on proposed indicators in this project will be used in a software program called the Conservation Action Planning Workbook, Version 5a (TNC 2007), which features an easy-to-use, menu-driven interface in Microsoft Excel. As an ecological scorecard (*sensu* Stem et al. 2005), the Workbook is useful for multi-metric analyses. The indicator ratings are combined as a weighted average (weighting factors of Poor = 1.0, Fair = 2.5, Good = 3.5, Very Good = 4) score for each attribute and rolled up to an overall target "viability" score. If indicators fall within their acceptable range, then the KEA may be viewed as "Good", and by extension "Good" KEA status suggests desirable target status (Braun & Salafsky 2006).

The coarse-filter approach to monitoring accepts that for most ecosystems little is known about suitable and practical indicators, acceptable ranges of variation, and appropriate scales (Vora 1997). The Nature Conservancy‟s CAP is a highly iterative process that does not require "perfect" information (Braun & Salafsky 2006). Instead it forces the user to get comfortable with uncertainty and to move forward with first approximations (Tear et al. 2005). As knowledge and resources expand and the project advances, it is expected the prescriptions will be refined and improved. The idea is to gather the best available information up front, document key assumptions and uncertainties, and move forward with a willingness to adapt and backfill data gaps. Most of the model offered in this report should be transferable, either directly or following modification, to other pine barrens and sand plains of the region. Modifications are encouraged if site-specific or more local information exists (e.g., fire history record), or research to inform prescriptions is implemented.

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II. Size & Extent

- ✓ Habitat loss is a major driver of biodiversity loss worldwide (Fahrig 2001, Foley et al. 2005).

- ✓ The total habitat size or areal extent is an important although not decisive factor in supporting healthy ecological processes (e.g., fire regime) and biotic assemblages. Multi-scale studies suggest that pine barrens should be protected at multiple spatial scales, but because of human resource constraints the best scale is probably that with the greatest number of imperiled species (Grand & Cushman 2003, Grand et al. 2004).

- ✓ Conservation has shifted towards valuing larger spatial and organizational scales, but many landscapes afford little opportunity for conserving large areas (Schwartz 1999). The APB is similar to prairie reserves of the Midwest in that it supports many species and many rare species on very limited acreage (Panzer & Schwartz 1998, Barnes 2003). We may be trying to protect a smaller land base of remnant pine barrens than we would like, but as Schwartz (1999) points out "…*we ought to prefer the risk of losing diversity in small reserves over the guaranteed loss of diversity by neglect*".

- ✓ Coarse-filter planning based on the umbrella and focal-taxon concepts (Lambeck 1997, Roberge & Angelstam 2004) would recommend a reserve that accommodates species with the largest area requirements. However, this criterion may overlook localized variation and operate at scales beyond the limits of managed areas and human resources (Prendergast et al. 1993, Fleishman et al. 2001). Recent thinking on umbrella species in ecosystem-based conservation has phased out traditional area criteria (Fleishman et al. 2000, Roberge & Angelstam 2004, Bried et al. 2007).

- ✓ Ecologically speaking, APB natural communities are best viewed as dynamic, shifting mosaics of successional types rather than discrete patches or ecological units. From a management and monitoring logistical standpoint, however, it is necessary to recognize discrete sampling areas on the landscape. Spatial and sampling scales at which inferences are drawn (plot vs. stand vs. patch vs. patch complex vs. reserve) place major challenges on establishing thresholds.

- ✓ The proposed size/extent attributes are habitat amount, patch size, core area, and suitable Karner blue butterfly habitat.

Habitat amount *(total pine barrens area)*

Rationale: The importance of overall amount and quality of natural land cover across a landscape can not be overstated (Fahrig 2001, Lindenmayer et al. 2008). Taxa ranging from mammals and birds to insects and plants are all highly sensitive to the areal extent of landscapes and habitat fragments (e.g., Burbidge et al. 1997, Renjifo 1999, Laurance et al. 2002 and references therein). Holding other factors constant, bigger is always better when it comes to total amount of reserve and valuing areas for protection (Diamond 1975, Noss 1987, Schwartz 1999). Larger reserves may accommodate species with larger area requirements, have less border and edge effect, and support larger populations with lower extinction probabilities (Schwartz 1999). Of course the "more is better" paradigm does not necessarily translate to "small is bad", as noted by Schwartz (1999).

Ideally the reserve design should encompass a minimum dynamic area, or the smallest area with a natural disturbance regime to support and sustain native biodiversity (Pickett & Thompson 1978). The minimum dynamic area should be many times larger (e.g., 50–100×) than the largest expected disturbances if the landscape is to be maintained in dynamic equilibrium (Shugart & West 1981). Minimum dynamic areas identified for grasslands of the Great Plains were all greater than 1,000 km^2 (Samson et al. 2004). The obvious limitation with the minimum dynamic area concept is the realism of attaining large areas when reserves are typically small and habitat is dwindling (Noss & Harris 1986). And non-equilibrium dynamics are probably the norm in ephemeral habitat like pine barrens.

Reserve size by itself may not predict population size or diversity, and the area of a particular land cover type will rarely reflect the amount of suitable habitat for a given species (Schwartz 1999, Lindenmayer et al. 2008, Schlossberg & King 2009). An analysis of flora species-area relationships in South Africa concluded that reserves of only 4–15 ha minimum are needed to avoid species losses (Cowling & Bond 1991). Size is not all that matters, therefore small sites should not be written off as unworthy of protection.

The Albany Pine Bush landscape historically spanned over 10,000 ha of sandy soils (Barnes 2003) and was one of the largest inland areas of pine barrens vegetation in the glaciated northeastern United States (Gebauer et al. 1996). By the late 1980s, less than 10% of the local pine barrens remained in the APB study area (Givnish et al. 1988). However, since that time and the formation of the Albany Pine Bush Preserve Commission in 1988, strategic land acquisition and ecological management have halted and reversed the trend. The most well reasoned hypothetical minimum viable area (Good rating) of APB pine barrens communities is 2,000 fire-manageable acres (see Givnish et al. 1988). "Fire-manageable" refers to existing pine barrens plus three land cover types (open fields, oak-pine forest, bare ground) that can be converted to pine barrens with the assistance of prescribed fire. Three additional cover types are not directly fire-manageable but are restorable via forest clearing and general site preparation: white pine (*Pinus strobus*) forest, black locust clones, and native aspen clones. The remaining cover types in the APB are not fire-manageable or restorable (lawn, developed, open water), or are desirable wetlands like woodland vernal pools and pine barrens vernal ponds (Bried & Edinger 2009).

Based on GIS land cover analysis in May 2003, pine barrens communities in the ~12,000-acre study area covered roughly 1,900 acres (B. Kinal, *unpublished data*). Only about 1,000 of these acres are Commission-owned and therefore manageable, but the Commission also owns ~1,600 acres of fire-manageable or otherwise restorable land cover other than pine barrens. This means the viable 2,000 acre benchmark is already achieved, if one assumes that at least two-thirds of the 1,600 restorable acres will be converted to pine barrens in the future. A viable preserve goal of ≥2,000 manageable acres meets or exceeds the home ranges of at least some of the more area-demanding birds and mammals in New England (DeGraaf & Yamasaki 2001). The Poor-Fair threshold of 1,000 acres is arbitrary. The desired cover of >8,500 acres is the estimated amount of current and restorable pine barrens in the APB study area. This lofty goal will motivate continued strong efforts to expand and buffer core pine barrens in the landscape. Projected land acquisition and long-term restoration potential make the Very Good threshold seem attainable.

Indicator: *current and restorable total acreage of pitch pine-scrub oak barrens, thickets, and forests*
Poor <1000
Fair 1000-1999
Good 2000-8500
Very Good >8500

Limitations

- ✓ The 2,000 acre mark put forth by Givnish et al. (1988) and discussed during the 2004 planning workshop seems reasonable for defining the reserve‟s contemporary minimum size, albeit probably not its minimum dynamic area.
- ✓ The Very Good level is far below the estimated historical extent (>25,000 acres) of pine barrens in this landscape (Barnes 2003).
- ✓ A central theme in conservation biology through the 1990s and present has been designing reserve *networks* to maximally represent species diversity (Cabeza & Moilanen 2001, Rodrigues & Gaston 2002, Possingham et al. 2006). The APB, like other scattered remnants of pine barrens, is an isolated reserve in an urbanized setting that affords little to no opportunity for networks.

Indicator: *pine barrens area expressed as percentage of the APB study area*
Poor <10%
Fair 10–30%
Good 30–50%
Very Good >50%

Effects of habitat fragmentation are likely to be revealed when habitat coverage drops below 50% of landscape area (Flather & Bevers 2002). Major ecological change or threshold response in land cover should occur when area declines to approximately 20% (±10%) of the landscape, this based on reviews of modeling simulations and patch-level studies of birds and mammals (Andrén 1994, Fahrig 1998). Radford et al. (2005) supported this theory by finding a 10% threshold for woodland bird species richness. The thresholds in the second indicator were set according to this information. By including a „fragmentation threshold", the proposed indicator measurement accounts for interaction between fragmentation and habitat loss. Present area of APB pitch pine-scrub oak barrens, thicket, and forest needs to approximately double to achieve the „Good" rating of 30–50%.

Patch size

Rationale: Mean patch size is frequently used to characterize landscape structure and may affect not only species richness but also local extinction and turnover rates (Boulinier et al. 2001). Small patches generally contain fewer species than large patches (Debinski & Holt 2000). Many species, especially large animals, will disappear from habitat fragments that are reduced to areas smaller than the minimum required home ranges or territories (Wilcove et al. 1986, Saunders et al. 1991). Birds and mammals have relatively large territorial requirements and thus should be more area-sensitive than other pine barrens taxa. Ovenbird, white-breasted nuthatch, black-and-white warbler, scarlet tanager, hairy woodpecker, red-eyed vireo, wood thrush, and great crested flycatcher have been viewed as area-sensitive species (Forman et al. 1976, Robbins et al. 1989). A study in southeastern Massachusetts pine barrens reported mean territory sizes of 0.64 ± 0.15 ha in rufous-sided towhees, 0.69 ± 0.15 ha in common yellowthroats, and 0.89 ± 0.37 ha in prairie warblers (Morimoto & Wasserman 1991); all of these species have been documented in or near the APB (Barnes 2003). Habitat area requirements and home ranges are much greater than territory sizes and should therefore be the focus of animal-based area threshold setting.

Vickery et al. (1994) evaluated occupancy patterns of early-successional birds breeding across a broad size gradient (90 sites of 0.3 to 404 ha) of grassland-barren sites in coastal Maine. Six of their analyzed species are documented in the APB, including a Species of Greatest Conservation Need (brown thrasher). Brown thrasher, common yellowthroat, and song sparrow ("edge species") showed the only negative incidence trends with patch area, presumably because the relative amount of shrubby habitat diminished with increasing area and habitat management intensity. Area requirements of grassland species ranged from about 10 to 200 ha, and the authors recommended protecting at least 50 ha (preferably 100–200 ha) of contiguous grassland for rare birds.

Among large carnivores known from the APB (see Barnes 2003), bobcats may require ~3,000 contiguous hectares, fishers ~2,600 ha, and foxes (gray and red) ~400 ha (Gittleman & Harvey 1982). In New England red foxes may travel over distances of 15 to 20 miles and occupy home ranges of at least 1,400 ha (Harrison et al. 1989). Fisher home range was estimated at 1,920 ha in New Hampshire and at almost twice this area for males in Maine (Kelly 1977, Arthur et al. 1989). In the APB, a female fisher used approximately 800 ha and foxes covered around 400 ha (R. Kays, Curator of Mammals, New York State Museum, *personal communication*). Home range of the coyote in the APB is approximately 575–680 ha (Bogan 2004); estimates are much larger elsewhere in the region (DeGraaf & Yamasaki 2001). In urbanizing shrub habitat of coastal southern California (Crooks 2002), 180 ha fragment size was the estimated cutoff for 50% probability of bobcat occurrence (bobcats are rare in the APB); the probability fell to zero at about 100 ha and rose to 100% at about 500 ha. In the same study, long-tailed weasel (resident of the APB) showed a lower probability of occurrence and lower relative abundance per unit area in smaller and more isolated habitat patches. In contrast to these species, probability of occurrence for domestic cats, a potential nuisance in the APB (Kays & DeWan 2004), dropped below 50% in fragments larger than ~140 ha.

Indicator: *mean patch area (acres)*
Poor <125
Fair 125-349
Good 350-1200
Very Good >1200

The Poor-Fair threshold (125 acres) is the minimum estimated habitat for protecting grassland-barrens birds in coastal Maine (Vickery et al. 1994). The Fair-Good threshold of 350 acres will more likely exclude than promote domestic cats (Kays & DeWan 2004), and achieving the Good-Very Good threshold of 1,200 acres may attract the bobcat, coyote, and red fox (Gittleman & Harvey 1982, DeGraaf & Yamasaki 2001, Crooks 2002, Barnes 2003, Bogan 2004); the red fox may strongly defend much smaller territories (Barnes 2003). Thresholds of Poor-Fair = 150 acres and Fair-Good = 300 acres were proposed during the 2004 planning workshop.

Indicator: *smallest patch (acres)*
Poor <25
Fair 25-124
Good 125-350
Very Good >350

The Fair-Good and Good-Very Good thresholds are shifted up one category from the previous indicator and correspond to home range sizes for some APB mammals (e.g., striped skunk; DeGraaf & Yamasaki 2001). The Poor-Fair threshold (25 acres) generally corresponds to requirements of relatively area-sensitive shrubland birds like golden-winged warbler and yellow-breasted chat (Dettmers 2003). Also, the long-tailed weasel needs home range space of at least 25 acres (DeGraaf & Yamasaki 2001).

Limitations

- ✓ "Patch" is a human construct that might not be particularly meaningful for some taxa or species assemblages (Lindenmayer et al. 2008).
- ✓ Patch context, or the nature of the landscape surrounding a patch (i.e., the "matrix"), functionally modifies raw patch area in complex ways (Fahrig 2001).
- ✓ Single or average patch size may not effectively capture the role of patch ensembles (Bennett et al. 2006), and mosaics of different patches with varying burn schedules are especially important in fire landscapes (Parr & Andersen 2006).
- ✓ The proposed attribute includes a measure of smallest patch, but size of the largest patch and its total perimeter can be an important structural attribute in patchy landscapes (e.g., Flather & Bevers 2002).

- ✓ Bird and mammal species differ in their preferences for amounts of forest versus non-forested habitat – the current indicator thresholds do not distinguish between the amounts of pitch pine vs. scrub oak dominated area.

Core area

Rationale: A core area is free of perceived edge effects and so represents area of high conservation value on the landscape (Grand et al. 2004, Beazley et al. 2005, Wei & Hoganson 2005). The „core-area model" predicts the quantity of interior habitat that is free from edge effects within fragmented reserves (Laurance & Yensen 1991). In some cases estimating core habitat provides key insight into species" movement behavior and may be superior to other edge-related measures like fractal dimension, shape index, or perimeter/area ratio (Stamps et al. 1987, Groom & Schumaker 1990).

Grand and Mello (2004) suggested conserving 300–600 m radii core areas for rare moths, which converts to about 28–113 ha (69–279 acres). Parkes et al. (2003) suggested that remnant blocks of vegetation should exceed about 50 acres with core areas of at least 124 acres. This vegetation core area and the midpoint of the previously stated moth-based range (174 acres) are roughly similar.

A recent study in a Massachusetts" pine barrens looked at moth and bird abundance in relation to numerous habitat types and scales (18, 70, 280, 630, and 1,120 acre circles) (Grand et al. 2004). Several of the species from that study are found in the APB and listed as Species of Greatest Conservation Need in New York, including whip-poor-will, prairie warbler, scarlet tanager, brown thrasher, barrens daggermoth, and barrens buckmoth. Scrub oak was a significant predictor of bird abundance but scale relationships varied by species. Numbers of whip-poor-will were positively related to scrub oak frost pockets at the 70-acre scale, prairie warbler to scrub oak areas at the 280-acre scale, scarlet tanager to scrub oak or mixed woods dominated landscapes at the 1,120-acre scale, and brown thrasher to scrub oak dominated landscapes at the 630-acre scale. Hairy woodpecker was positively associated with pitch pine-scrub oak thicket at the 18-acre scale, whereas scarlet tanager was negatively associated with this community at the 630-acre scale. The barrens buckmoth and pine barrens itame preferred scrub oak dominated land cover (including thicket) at 280- and 630-acre scales, respectively; scrub oak serves as the larval food source to both species (Wagner et al. 2003). None of the 17 bird species or six moth species analyzed by Grand et al. (2004) that also occur in the APB showed significant associations to pitch pine-scrub oak forest.

A threshold of 25 acres may minimize nest predation rates and provide enough shrubland for area-sensitive species like the golden-winged warbler, prairie warbler, and yellow-breasted chat (reviews by Patton 1994, Dettmers 2003). As a group, however, early successional or shrubland-breeding birds appear relatively insensitive to patch size (Dettmers 2003), and in fact may prefer smaller sites with more edge (Vickery et al. 1994, Woodward et al. 2001; but see Schlossberg & King 2008).

Combining the above information, thresholds of 70, 150, and 280 acres seem like reasonable first approximations for Poor-Fair, Fair-Good, and Good-Very Good core area ratings, respectively. Doubling these thresholds yields roughly that proposed at the 2004 planning workshop for smallest patch (i.e., 150, 300, and 600 acres, respectively). The

Good-Very Good threshold corresponds to spatial scales of scrub oak barrens at which the barrens buckmoth, pine barrens itame, prairie warbler, and brown thrasher, all of conservation concern in the APB, significantly associated in southeastern Massachusetts (Grand et al. 2004).

Indicator: *total patch area minus the total edge effect zone*
Poor <70 acres
Fair 70-150 acres
Good 151-280 acres
Very Good >280 acres

The „total edge effect zone‟ includes 340 m from major roads (Rt. 155, Rt. 20, Interstate 95), 150 m from minor roads, 75 m from trails, and 50 m from residential property (see „Edge effects‟ narrative in III. Fragmentation & Edge Effects).

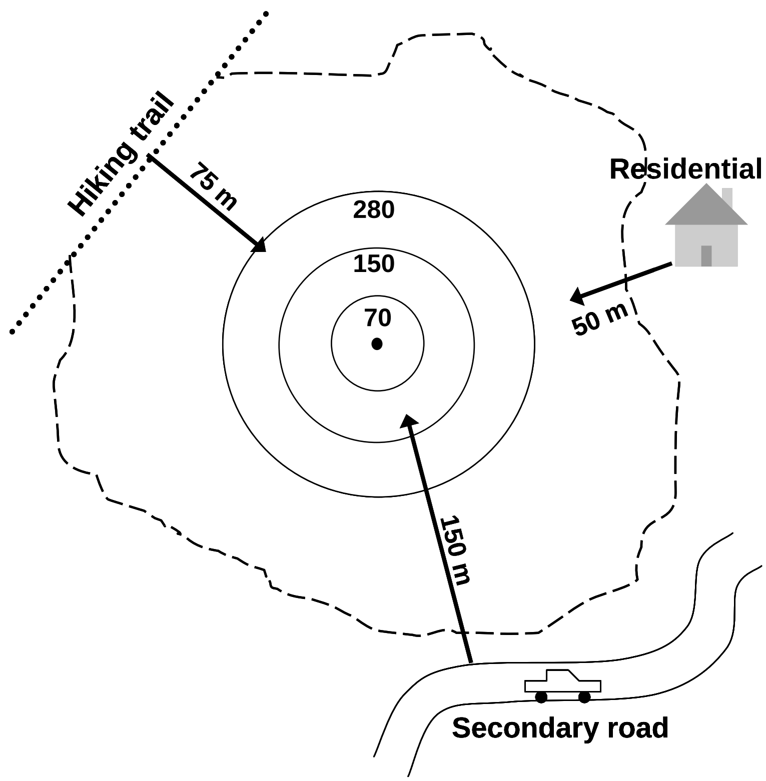
Indicator: *circular acreage (Πr^2) around the approximate patch center that is free of edge effects from roads, trails, or residential development*
Poor <70 acres
Fair 70-150 acres
Good 151-280 acres
Very Good >280 acres

The second indicator is a more conservative measure. The same thresholds of acreage and effect distance are used, but now the "effect-free" zone radiates out from the patch center (see figure below). It assumes that the deepest interior of the patch is least altered by adverse human activity and lies farthest from the matrix. The status of central core area is measured by the deepest line of penetration from the road, trail, or residential edge to the patch center. Thus, a patch with enough core area overall (i.e., 150+ acres) may not have enough central core area. The concept is analogous to the „multiple-use-module‟ for insulating core preserves with layers of buffer zones (Noss & Harris 1986, Noss 1987).

Limitations

- ✓ A number of potential edge effect features are ignored. This indicator will need to be modified if new research finds additional types of edge effects or suggests better estimates for trail, road, and residential effect distances.
- ✓ The ecologically-based ratings may not be achievable given the potential land base for APB pine barrens.

Hypothetical example of the „central core area" concept as controlled by fragmenting features. The dot is the approximate center of the pine barrens habitat patch (dashed line). Rings around the center represent the core area rating thresholds (Poor-Fair = 70 acres, Fair-Good = 150 acres, Good-Very Good = 280 acres). The arrows indicate the lines of deepest edge effect from a trail, road, and residential property (see „Edge effects" narrative in III. Fragmentation & Edge Effects). In this example, the deepest effect distance (aimed towards the center) comes from the road feature. The central core area of this patch would be rated as „Fair" (≥70 but ≤150 acres edge-free). Removing the road would bring the patch rating to „Good" (>150 but <280 acres edge-free) and removing the hiking trail would yield „Very Good" (>280 acres edge-free).



Suitable Karner blue butterfly habitat

Rationale: Prairie remnants in shrublands offer food plants required by rare Lepidoptera (Givnish et al. 1988, Wagner et al. 2003). The federally endangered Karner blue butterfly (KBB) (*Lycaeides melissa samuelis*) is a flagship species for the APB and a potential surrogate for maintaining and restoring prairie- or savannah-like habitat (Dirig 1994). Wild lupine abundance (*Lupinus perennis*; KBB larval host plant), fire return interval, and grassy shrubland or pine-oak savannah structure are probably key regulatory factors of KBB metapopulation viability. As such, prairie openings and, more generally, suitable KBB habitat is regarded as a fourth desirable successional variant of the pine barrens conservation target. Currently in the APB this butterfly occupies old fields, new fields converted from forest, powerline rights-of-way, and sand pits on the periphery of shrubland habitat. Thus one major APB goal is to facilitate expansion of KBB populations into pine barrens habitat through continued restoration and accelerated colonization (captive release).

The New York KBB recovery team has a detailed management and monitoring program in place for this species (Bried 2009, Tear et al. unpublished data). On the monitoring end the team has developed a detailed measures scheme of population and habitat (restoration) indicators specific to the species. One key recovery indicator is total amount of suitable habitat in each New York metapopulation recovery area, with "suitability" defined by lupine density, nectar diversity (richness, density, evenness), and physical structure (grass and overstory cover, shade heterogeneity). For patches to count as suitable they must score Good or better from the collective indicators. Additionally, each suitable patch must be at least 0.62 acres and belong to a subpopulation of at least 12.4 acres that is within 1 km of at least two other subpopulations. The rating scheme draws directly from recommendations in the KBB federal recovery plan (USFWS 2003).

Indicator: *Amount of suitable Karner blue habitat in the preserve*
Poor <160 acres
Fair 160-319 acres
Good 320-639 acres
Very Good ≥640 acres

SUMMARY TABLE

	Key ecological attribute	Indicator	Ratings: Poor	Fair	Good	Very Good
Size and Extent	Habitat amount	current + restorable total acreage	<1000	1000-1999	2000-8500	>8500
		target area expressed as percentage of APB study area	<10	10-30	30-50	>50
	Patch size	mean patch area (acres)	<125	125-349	350-1200	>1200
		smallest (acres) patch	<25	25-124	125-350	>350
	Core area	individual patch area minus the total edge effect zone (in acres)	<70	70–150	151–280	>280
		circular (Πr^2) edge-free area around the patch center (in acres)	<70	70–150	151–280	>280
	Suitable Karner blue butterfly habitat	amount (acres) of suitable Karner blue habitat	<160	160-320	320-640	>640

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III. Fragmentation & Edge Effects

- ✓ Habitat fragmentation, or the process of dividing contiguous habitat into smaller, isolated patches (Fahrig 2003), is one of the most productive areas of study in conservation biology (Fazey et al. 2005). Fragmentation may greatly exacerbate the negative biological effects of habitat loss alone (Saunders et al. 1991, Hanski & Ovaskainen 2000). However, whereas habitat loss nearly always results in fewer species, smaller populations, and increased risk of extinction (Diamond 1975), fragmentation may have positive effects on some species, such as "edge specialists".

- ✓ Fragmentation increases extinction risk (Wilcove 1987, Reed 2004), not only via direct effects like blocking dispersal but also by facilitating threats like exotic plant invasions (Lonsdale 1999, Schmidt & Whelan 1999, Laurance et al. 2002). This means effects of fragmentation are not always clear or linear, such as disruptions in insect pollinator services (Jennersten 1988), but rather may show changes only at extreme thresholds or peaks at intermediate levels of disturbance (Fahrig 2003).

- ✓ Fragmentation may limit contiguous area-demanding apex predators, leading to subsidized feeding by mesopredators (e.g., raccoons, domestic cats) and over-predation on native fauna (Crooks & Soule 1999, Odell & Knight 2001, Kays & DeWan 2004, Manley et al. 2006). Nest parasitism and edge-predation all tend to increase with level of fragmentation (Andren & Angelstam 1988, Paton 1994, Robinson et al. 1995, Schmidt & Whelan 1999), but edge species like robins, blue jays, and brown-headed cowbirds are favored by fragmenting features (Hickman 1990, Miller et al. 1998).

- ✓ Fragmentation can cause changes in environmental conditions along habitat margins or edges, popularly known as "edge effects" (Murcia 1995, Fagan et al. 1999). There are many different types of sharp boundaries ("edges") between patches and changes in biological and physical conditions across those boundaries (Ries et al. 2004, Harper et al. 2005), but there are few studies of edge effects in non-forested ecosystems (Lindenmayer et al. 2008).

- ✓ A series of studies in a southeastern Massachusetts pine barrens suggest that landscape level environmental factors may influence animal abundance and distribution more than finer spatial scales (Grand & Cushman 2003, Grand & Mello 2004, Grand et al. 2004). Also at the landscape scale, fragmenting features like roads may have significant negative effects on pine barrens plant diversity and recruitment (Gill 1997, Brosofske et al. 1999).

- ✓ It is ecologically tempting but impractical to conceptualize the APB landscape as a series of species-specific habitat gradients. Therefore, pine barrens conservation assessment and management must "cast a wide net" and base fragmentation thresholds on the most area-demanding or dispersal-limited rare animals (Lambeck 1997).

- ✓ Lindenmayer et al. (2008) observed that "*connectivity remains one of the most difficult areas of landscape conservation*" in terms of measurement and defining the appropriate scale. Several dozen measures of fragmentation are used in practice (McGarigal et al. 2002). These measures describe habitat configuration and not habitat loss per se (Fahrig 2003).

- ✓ Of the numerous potential fragmentation attributes (Noss 1999, McGarigal et al. 2002), some of the most reliable and simplest are suggested here: patchiness, patch isolation distance (nearest-neighbor connectivity), and perimeter/area ratio. A variety of edge effects (roads, trails, residential) are proposed since species sensitivity to edge may vary by edge type (e.g., Suarez et al. 1997). Conceptual foundation for the edge effect attribute is captured in the „multiple-use-module", a tiered strategy of reserve design calling for buffer habitats that insulate core areas from the developed landscape (Noss & Harris 1986).

Patchiness

Rationale: The process of fragmentation changes habitat configuration in part by increasing the number of patches (Fahrig 2003, Watling & Donnelly 2006). Despite the unresolved *SLOSS* debate for reserve networks (Diamond 1975, Soule & Simberloff 1986, Burkey 1989, Schwartz 1999), the proposed indicator assumes that a "single large" *patch* is better than "several small" *patches* in a single reserve. However, if habitat and biota are finely distributed over numerous small patches, then perhaps maximizing quality of existing patches and trying to prevent further fragmentation should be the focus. With this line of thought, current patchiness of the APB is used to anchor the rating scheme at „Good". At each monitoring event, the current patchiness resets to Good, regardless of whether it was Poor, Fair, or Very Good at the prior time step. The other category ratings follow the assumption that less patchiness is better. This does not mean that creating new patches of pine barrens would be undesirable, only that we do not want to fragment existing patches. The analyst can decide whether to rate increased patchiness as „Fair" or „Poor", and may instate rules like „Fair" will be any increase in current patchiness and „Poor" will be a doubling of current patchiness. In deciding between „Poor" and „Fair", it is advised that analysts try each option in turn to assess whether it may affect the overall pine barrens rating from all indicators combined.

Indicator: *number of target patches, where patches are delineated by obvious fragmenting features like roads*
Poor increase existing P
Fair increase existing P
Good maintained patchiness (P)
Very Good reduce P

Limitations

- ✓ We are assuming that "single large" is better than "several small" despite the unresolved *SLOSS* debate and its emphasis on reserve networks rather than single reserves (Soule & Simberloff 1986, Schwartz 1999). In contiguous habitat, disease and exotic species may spread more easily and catastrophes may have more lasting impacts.
- ✓ Landscape connectivity, or physical linkages of vegetation cover types in the landscape, ignores species-specific and functional linkages at multiple scales (Lindenmayer & Fischer 2007).

Patch isolation distance

Rationale: Distance between habitat patches is an important feature of landscape structure, equilibrium dynamics, and biodiversity loss (Boulinier et al. 2001, Fahrig 2003), and therefore a necessary component of reserve design and management (Schultz 1998). Connectivity will likely influence the regenerative capacity and long-term survival of remnant vegetation in the APB (Cunningham 2000). A variety of distances between same seral stages and same age-classes will more likely benefit dispersive species adapted to particular seral stages and/or age-classes (Givnish et al. 1988). Because of the naturally occurring spatial interspersion of target community types and age-classes in the APB, preserve management on multiple time and spatial scales will offer greatest niche variety and best promote a characteristic and diverse species mix.

Many factors control an animal‟s use of fragmented habitat, including corridors (physiognomy, length, width), stepping stones (amount, density, configuration, edge types), species perceptions, population density, and amount of contrast between matrix habitat and suitability of patches and corridors (Fahrig 2003). Corridors may alleviate isolation but their value, like response to fire, is species specific and often unclear (Clinchy 1997, Beier & Noss 1998). In general it is probably safe to assume that corridors will have positive or neutral effects in pine barrens landscapes like the APB.

Smaller animals tend to be more dispersal-limited than larger ones, so insects are a good focal taxon (Lambeck 1997) for thresholds. Research on temperate-centered Lepidoptera has led to a general recommendation that inter-patch distances should not exceed 1 km unless connected by corridors (Smallidge & Leopold 1997); corridors may enhance inter-patch movement of open-habitat butterfly species (Haddad & Baum 1999). As a U.S. federally endangered species, the Karner blue butterfly is a flagship for conservation and management in the APB, with flight capacity (average <1000 m dispersal, average <200 m lifetime movement) typical of other butterflies. Since a major conservation goal for the APB is to establish a viable Karner blue metapopulation in native barrens habitat, patches of pine barrens should be arranged within the typical maximum dispersal distance of this species (~1 km; USFWS 2003). The lower limit of the „Good‟ threshold is the state and federal cutoff for separating Karner blue subpopulations and helps ensure that occupied habitat is not clustered too tightly. The scorecard analyst should consider the amount of variation around point estimates (arithmetic mean) – the more distance variation the better (theoretical worst-case scenario is mean ± zero), consistent with the prediction that more heterogeneity will help maximize biodiversity in this landscape (Givnish et al. 1988).

Indicator: *mean nearest-neighbor distance (in kilometers) among target patches*
Poor >2 km (point estimate)
Fair 1–2 km (point estimate)
Good 0.2–1 km (contains point estimate but not 95% confidence limits)
Very Good 0.2–1 km (contains point estimate and 95% confidence limits)

Limitations

- ✓ Nearest-neighbor connectivity measures distance to a single patch, which may be unreliable (overly simplistic) because it ignores the proximity of other neighboring patches (Bender et al. 2003). Furthermore, animal perceptions of "distance" may vary with context, thus connectivity measures that account for animal mobility, patch size, and patch arrangement may be more meaningful. For example, when inter-patch distance exceeds movement capacity and transient habitat forms a linear unbroken strip through the matrix, "corridor distance" may prevail. But when inter-patch distance is less than movement capacity and transient habitat is a network of small patches, "stepping stone distance" may prevail (Haddad 2000).
- ✓ Patch isolation distance within the APB landscape obviously ignores the landscape"s connectivity to other natural areas. A fundamental tenet of island biogeography and predicting how many species a preserve can hold is how connected that preserve is to other preserves (Diamond 1975).
- ✓ Patch isolation is not so much a measure of habitat fragmentation as it is a measure of the lack of habitat in the landscape surrounding the patch (Fahrig 2003). The monitoring scheme may need an explicit matrix effect, taking into account barrier features that may place serious constraints on even the smallest inter-patch distances (see Laurance et al. 2002).
- ✓ Area-based isolation metrics (e.g., proximity index) and accounting for factors like patch shape and matrix hostility may work better at predicting animal dispersal than nearest-neighbor distance (Bender et al. 2003).
- ✓ Amphibians and reptiles may be particularly sensitive to pine barrens fragmentation and may demand more conservative distance thresholds than flying insects.

Perimeter/area ratio

Rationale: Efforts at reserve design continue to emphasize overall size and shape, as island biogeography theory intended (MacArthur & Wilson 1967). Long thin reserves have higher edge-to-area ratio (less core area; Laurence & Yensen 1991) and are more sensitive to edge effects and weed/pest invasions (Fagan et al. 1999), thus compact or circular dimensions are preferred.

A reasonable goal, both ecologically and operationally, is to minimize the overall reserve boundary length. A circle is the most compact shape possible, so it makes sense to weigh boundary length against this theoretical minimum perimeter/area ratio. The suggested measure is the ratio of the total patch boundary length to the circumference of a circle with the same area as the patches combined (Possingham et al. 2000):

$$\frac{boundary\ length}{2\sqrt{\pi \times area}}$$

The higher the ratio the more fragmented the reserve; values approaching 1 indicate increasingly compact or clustered pine barrens area (i.e., approaching the shape of a circle).

Indicator: *ratio of total patch boundary length to the theoretical minimum perimeter/area ratio (R)*
Poor increase R
Fair increase R
Good ≤ current R
Very Good 1 (theoretical)

The „Good‟ rating of „≤ current R‟ defines a goal of maintaining or improving the current R. The analyst can decide whether raising R deserves a „Fair‟ or „Poor‟ rating. For example, less than 25% increase from current R might be taken as „Fair‟ and more than 25% increase might be taken as „Poor‟. As with „patchiness‟, it is advised that the analyst try the „Fair‟ and „Poor‟ options in turn to assess whether it may affect the overall pine barrens rating (i.e., all indicators combined).

Edge effects

This section builds a rationale for edge effect distances off roads, trails, and residential property. Thresholds were incorporated into the „Core area‟ attribute for Size & Extent, but are more appropriately explained as part of Fragmentation & Edge Effects to distinguish the effects from habitat loss per se (reviewed in Fahrig 2003).

Roads

Roads may divide metapopulations, reducing gene flow and creating less stable and more vulnerable subpopulations (Mader 1984, Wiens 1996). Noise, visual stimuli, pollution, direct mortality, and movement inhibition are some of the many adverse consequences of roads. Highway noise, for example, interferes with reproductive vocal communication in birds and makes it more difficult for deer to detect predators (Reijnen et al. 1995, Forman & Deblinger 2000). Landscapes with extensive roads and/or high traffic volume will interfere with complex movement behavior in herpetofauna (Fahrig et al. 1995, Findlay & Houlahan 1997, Hels & Buchwald 2001, Houlahan & Findlay 2003, Cushman 2006, Eigenbrod et al. 2008, Shepard et al. 2008).

Reijnen et al. (1995) studied the zone of influence around roads and found lower breeding bird densities closer to roads than farther away. They measured noise loads and visibility of cars in deciduous and coniferous woodland types throughout the Netherlands.

Traffic density in their deciduous study areas ranged from 8,000 to 61,000 cars per day, and in coniferous study areas from 29,000 to 69,000 cars per day. Approximately 25% of 41 species showed significantly lower densities near roads, with noise having a stronger effect than visibility. Effect distances (distance from road to point of significantly reduced bird population density) varied greatly by species ranging from 40 to 1,500 m and 70 to 2,800 m at 10,000 and 60,000 cars per day, respectively, in deciduous forest, and from 50 to 790 m and 100 to 1,750 m at these car densities in coniferous forest.

Forman & Deblinger (2000) studied the "road-effect zone" along a 25 km stretch of a four-lane divided highway in the outer and middle suburbs west of Boston, Massachusetts. They estimated key road effects coming from stream alteration and wetland drainage, road salt, planted roadside exotics, moose and salamander corridor blockage, habitat avoidance by forest and grassland birds, and roadkills of deer. The estimated road-effect zone extended at least 100 m out for plant invasions to hundreds of meters and several kilometers out for road salt contamination, traffic noise interference of bird communication, and disruption of habitat suitability and travel corridors for large mammals. Combining all these factors the authors estimated a mean ecological effect distance of about 300 m from the edge of the road surface, with the area affected being about 0.6 km^2 km^{-1} of road length.

Traffic volumes on major roads in the APB fall within range of the Dutch and USA studies (Reijnen et al. 1995, Forman & Deblinger 2000). Annual average daily traffic (AADT) estimates along Rt. 155 from Rt. 20 to Washington Avenue Extension were 20,970 (in year 2005), 20,630 (2001), 20,390 (1998), and 20,100 (1995) vehicles per day. Estimates from Washington Avenue Extension to Rt. 5 were 28,040 (2004), 18,320 (2000), and 25,090 (1997) vehicles per day. Along Rt. 20 from Rt. 146 (at the Stewart's Shop) to Rt. 155, AADT estimates were 41,870 (2005), 28,340 (1999), 25,310 (1996), and 25,950 (1995) vehicles per day. Along Interstate 90, including a 6.5 mile stretch west from the I90/I87 interchange, AADT estimates were 57,545 (2005), 62,550 (2004), and 60,740 (2002) vehicles per day (data source: *2005 Traffic Data Report for New York State*, New York State Department of Transportation). The high traffic volume on these roads is similar to the situation studied in Reijnen et al. (1995) and Forman & Deblinger (2000).

A conservative buffer width was set based on the upper range of effect distances observed for a volume of 60,000 cars per day. Because many taxa in the Dutch study do not overlap with APB taxa, except for a few genera (*Buteo* hawks, *Parus* titmice, *Scolopax* woodcocks, *Troglodytes* wrens), the median value in the confidence range reported for all species combined (11 in deciduous woodland, 5 in coniferous woodland) was used. Medians at the 60,000 car density were averaged to a 340 m effect distance from Rt. 155, Rt. 20, and I95, and medians at the 10,000 car density were averaged to a 150 m effect distance from minor roads. This effect range includes the smaller effect distances and the rough overall effect (~300 m) reported in the Massachusetts study.

Indicator for core area: mean area of target beyond 340 m from major roads (Rt. 155, Rt. 20, I95) and 150 m from minor roads (all other paved travel corridors)

Limitations

- ✓ Traffic flows and thus noise loads, etc vary by road across the APB. Reijnen et al. (1995) modeled fixed car speeds of 120 km/hr (75 miles/hr), thus the effect distances might be relevant to I90 but are less applicable to smaller roads. Additionally, their study area had ~70% woodland cover adjacent to the road system, whereas adjacent woodland cover varies across the vast APB road network.
- ✓ Population density of the most sensitive forest-interior and grassland bird species may be reduced out to a kilometer from main roads (Reijnen et al. 1995). This suggests that the 340 m distance may be too short. In Massachusetts, six years (1993–1998) of breeding-season records for bobolinks and meadowlarks suggest that breeding is less likely or more irregular at sites within approximately 1 km of main roads (Forman & Deblinger 2000). Also, the scale of road effects for amphibians breeding in vernal pools and pine barrens vernal ponds may be much larger (e.g., >500 m scale of effect; Vos & Chardon 1998, Eigenbrod et al. 2008) than the current thresholds allow.
- ✓ Road ecology research has focused mainly on traffic and noise emissions, but some animals are deterred by the road surface itself (McGregor et al. 2008), thus measures of road density (Vos & Chardon 1998, Rytwinski & Fahrig 2007) may be informative in landscape viability assessment.
- ✓ Forman & Deblinger (2000) stress that the road effect zone is highly asymmetric; the proposed indicator assumes a symmetric effect.

Trails

Human recreation may accelerate the decline of animal diversity and populations (Garber & Burger 1995, Reed & Merenlender 2008). The draft recreation management plan for the APB acknowledges the negative effects of trails, both in their direct use (by hikers, etc) and as conduits for plant invasions and barriers to animal movement. Trail area of influence on wildlife can be significant even for seemingly benign activity, such as hiking (Gutzwiller et al. 1994, Taylor & Knight 2003).

Ideally, at least 150 acres of pine barrens should remain after subtracting a 75 m edge effect on both sides of trails. This effect distance is drawn from the estimated area of influence of passive recreation on breeding birds, small mammals, and ungulates (Miller et al. 1998, Taylor & Knight 2003, Lenth et al. 2008).

Indicator for core area: mean area of target beyond 75 m from hiking trails and fire breaks

Limitations

- ✓ Intensity and type of recreation varies by trail in the APB, but since no data are available to differentiate the effect, all trails are treated with equal weight.
- ✓ Location of the trail through the patch is ignored. For example, a trail that bisects a patch and leaves behind 300 acres (after subtracting the zone of influence) may have stronger fragmentation effects than if the trail ran closer or tangential to the patch edge.

Residential
Some edge effects are specific to residential development, such as domesticated animal disturbance. Not surprisingly domestic cat density is controlled more by human density than prey density (Sims et al. 2008), thus in urban settings these animals may pose a serious threat. The strongest evidence for edge effects in terms of bird depredation is for distances <50 m (reviewed by Paton 1994). For example, human-sensitive bird species showed lower densities 30 m compared to 180 m from houses in a Colorado shrub-oak community (Odell & Knight 2001). In the APB, the vast majority of house cat activity occurs within 50 m of the home residence (Kays & DeWan 2004). This was also the midpoint of an edge effect range shown to lower pitch pine seedling growth and survivorship in the APB (Gill 1997).

Indicator for core area: mean area of target beyond 50 m from residential edge

Limitations
- ✓ The 50 m effect distance has general and APB-specific support (Patton 1994, Kays & DeWan 2004), but nesting site patterns may be species-specific within this distance and one should not assume that nest predation decreases monotonically with distance (Woodward et al. 2001).
- ✓ The 50 m effect distance may not buffer against other potential disturbances (to wild animals) of residential development, like noise and light pollution.

SUMMARY TABLE

<table>
<tr><th rowspan="2"></th><th rowspan="2">Key ecological attribute</th><th rowspan="2">Indicator</th><th colspan="4">Ratings</th></tr>
<tr><th>Poor</th><th>Fair</th><th>Good</th><th>Very Good</th></tr>
<tr><td rowspan="6">Fragmentation and Edge Effects</td><td>Patchiness (P)</td><td>number of patches</td><td>increase P</td><td>increase P</td><td>current P</td><td>reduce P</td></tr>
<tr><td>Patch isolation distance</td><td>mean nearest-neighbor distance (km) among target patches</td><td>>2</td><td>≤2</td><td>≤1</td><td>NA</td></tr>
<tr><td>Perimeter/area ratio (R)</td><td>boundary length ÷ [2×√(π×area)]</td><td>increase R</td><td>increase R</td><td>≤ current R</td><td>1 (theoretical)</td></tr>
<tr><td>Road effect zone</td><td>mean area (acres) of target beyond 340 m from major roads and 150 m from minor roads</td><td colspan="4" rowspan="3">used in „Core area“ attribute</td></tr>
<tr><td>Trail effect zone</td><td>mean area (acres) of target beyond 100 m from hiking trails and fire breaks</td></tr>
<tr><td>Residential effect zone</td><td>mean area (acres) of target beyond 50 m from residential edge</td></tr>
</table>

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IV. Prescribed Fire Regime

"*Pyrodiversity begets biodiversity*"
– Martin & Sapsis (1992)

- ✓ An intact "natural" disturbance regime is critical to ecosystem-based conservation (Groves et al. 2002, Bengtsson et al. 2003). Pine barrens of the northeastern U.S. are a prime example of habitat dependant on disturbance, and active management is needed to protect the species relying on these systems (Litvaitis 2003). Reserves like the APB should be managed as a dynamic rather than static system (*sensu* Bengtsson et al. 2003) where natural disturbance regimes are simulated on a landscape scale.

- ✓ It is generally held that more fire regime patchiness and heterogeneity (e.g., patch mosaic burning) is better in fire-prone conservation areas (Parr & Andersen 2006). Indeed, no single management regime will benefit all species (Swengel 1998, 2001).

- ✓ The pine barrens landscape is homogeneous with respect to *type* of disturbance: fire is the key driver of pine barrens vegetation (Brosofske et al. 1999). However, acute fire effects vary among species and depend on such fire characteristics as flame lengths, severity, and frequency, and environmental conditions like fuel type/amount, temperature, weather, and season (Jordan et al. 2003).

- ✓ Fire is not the only disturbance facilitating early succession: land clearing for agriculture has helped create and shape the distribution of northeastern U.S. shrublands (Lorimer & White 2003). Motzkin et al. (2002) postulate that traditional agricultural practices like sheep grazing "…*may achieve many ecological objectives that are similar to prescribed summer burns*".

- ✓ Historical-geographical science is critical to contemporary understanding of natural systems (Foster 2002), and provides important perspective in guiding ecological management towards goals to restore and maintain natural ranges of variation (Aplet & Keeton 1999, Landres et al. 1999, Swetnam et al. 1999). Current ecological patterns and processes in northeastern U.S. pine-oak barrens are highly modified from that which occurred historically (Lorimer & White 2003), confusing our interpretation of historical vs. contemporary context. For example, centuries of woody species suppression by New England farmers may profoundly affect modern and future sand plains development (Motzkin et al. 1996, 2002). In the APB, fire-driven successional changes likely interact with variation in surficial deposits and historical land use disturbance (Gebauer et al. 1996, Finton 1998). Despite the complexity, historical ecologists advise that past and present dynamics must both be appreciated to effectively manage pine-oak barrens and achieve particular thresholds (Motzkin et al. 1999).

- ✓ There are two big logistical problems with relying on historical fire regimes: (1) fire records tend to be sparse and fragmented, and (2) landscapes like the APB are situated in densely populated areas where concern for human safety outstrips concern for nature. The latter problem is a direct result of rapid and widespread human population growth through the 20th century (Fahey & Reiners 1981, Motzkin et al. 1996).

- ✓ Many landscapes have been altered to points where historic rules are no longer appropriate or achievable (Lindenmayer et al. 2008), thus a paradigm shift towards embracing and promoting resilience of novel systems may warrant consideration (Egan & Howell 2001, Seastedt et al. 2008). Furthermore, the general emphasis on natural ranges of variability and viability as a basis for management (Aplet & Keeton 1999, Landres et al. 1999) may need balanced appreciation of cultural practices as environmental drivers in historically modified landscapes (Swetnam et al. 1999).

- ✓ Periodic fire sustains key components of the system while reducing fuel loads and chances for catastrophic wildfire. The following fire regime attributes (refugia, size, return interval, seasonality) are consistent with those selected by Noss (1999) for monitoring natural fire suppression in forests. They also overlap with the comprehensive fire management program in Kruger National Park, which recommends setting "thresholds of concern" for total percentage of area burned, desired patch-size frequency distribution, and seasonal fire distribution (Parr & Andersen 2006).

Refugia

Rationale: To drive the ecosystem renewal cycle there must be areas of the landscape untouched by disturbance (Holling 1986, Bengtsson et al. 2003). Portions of contiguous habitat should remain untouched during a fire season to allow refugia and post-fire community assembly (Schultz & Crone 1998, Harper et al. 2000). Moderate intensity, patchy fires at 5–10 year returns are likely to leave refugia for fire-sensitive plants and insects while still occurring frequently enough to reduce fuel loads that feed severe fires.

Many groups of specialized organisms benefit from permanent non-fire refugia or patchy fires that leave unburned refugia (Harper et al. 2000, Panzer 2003, Swengel & Swengel 2006). Swengel (1996) recommended sparing 80% of sites in grassland management areas to promote Lepidoptera. Panzer (2002) endorsed this threshold as a rule of thumb in managing for prairie insects. It should be noted, however, that unburned matrix quality (e.g., dead wood availability) and not simply *amount* of refugia may be a major limiting factor to pyrophilous insect populations (Saint-Germain et al. 2008).

Some butterflies may respond more favorably (higher abundance) to occasional large wildfire than to rotational prescribed burning (Swengel 1998), and several lycaenid species appear to be negative responders even when prescriptions mimic known historical regimes (New 1993). Schultz & Crone (1998) modeled effects of prescribed fires on persistence of the Fender"s blue butterfly in Oregon. Their simulation included return intervals of 1–5 years with 12.5–50% of patch area burnt in one fire. They recommended burning one-third of the butterfly"s habitat every year or every two years.

Using the above studies as a guide (Swengel 1996, Schultz & Crone 1998), the proposed ratings assume that one-third burned and two-thirds unburned habitat, for individual patches or across the target as a whole, is ideal (Very Good). The other ratings are gradual departures from the ideal range.

Indicator: *Seasonal amount (%) of spared (unburned) habitat*
Poor <25 or >90
Fair 25-50 or 80-90
Good 50-60 or 70-80
Very Good 60-70

Individual fire size

Rationale: A survey of historical fire records by The Nature Conservancy (as cited in Givnish et al. 1988) suggests that individual fires burned as much as 1,200 acres in the APB; Zaremba et al. (1991) reference a fire that burned thousands of APB acres in 1854. From 1968 to 1987, fires burned a cumulative total of 3,590 acres over 2,500 acres of the APB study area, but 22 of the 33 fires reported during this time were only an acre or less. The remaining 11 "large" fires averaged 326 acres (95% C.I. = 140 to 554 acres based on 10,000 bootstrap replicates). Unfortunately, these are small sample sizes and the accounts do not make it clear whether the fires were naturally ignited and self-regulated.

In contrast to APB, the Long Island Central Pine Barrens core area has 144 unambiguous fire size estimates dating back to 1931 (Jordan et al. 2003), the Shawangunk Mountains record includes 107 fires from 1842 to 1989 (Hubbs 1995), and the New Jersey Pinelands record includes annual average fire size from 1906 through 1976 (Forman & Boerner 1981). It seems more prudent to base thresholds on the detailed records from these other sites than use the sparsely documented local fire history. The Montague Sand Plain in Massachusetts is closest to the APB and has a fire incidence record from 1928 to 1994, but size is reported for only 15 fires (Motzkin et al. 1996).

Between 1931 and 1994 in the 12,503 acre core area of the Long Island Central Pine Barrens, average fire size was 365.7 acres and actual fires were expected within a range of 199.8 to 622.5 acres 95% of the time (based on 10,000 bootstrap replicates). Excluding one extremely large fire (burned 15,000 acres) as an outlier, the average fire size drops to 263.4 acres with the 95% error margin reduced to 186.8–369.2 acres. Adjusting these fire sizes to the spatial extent of current Commission-owned lands in the APB, via:

$\sum_{i=1}^{N}\left(F_i \div 12{,}503 \times 3{,}010\right) \div N$, where F is the size of the i^{th} fire and N = 143 fires we might expect an average fire size of 63.4 acres and the range 44.6 to 89.7 acres to contain actual fire sizes 95% of the time. This might be viewed as the acceptable range („Good"), meaning annual mean fire size should fall within this range. Assuming the most ecologically productive or at least typical fire sizes lie closer to the center of the range rather than near the limits, the desirable range (Very Good) was set at:

$$\frac{89.7+44.6}{2} \pm \frac{89.7-44.6}{4} = 67.2 \pm 11.3 = [55.9\,, 78.5]$$

Thresholds for Good, Fair, and Poor were set by progressively adding and subtracting 11.3 starting from the Very Good limits, ending at <33.2 or >101.1 acre Poor sized fires.

Indicator: *Long Island-based annual mean individual prescribed fire size (acres)*
Poor <33.2 or >101.1
Fair 33.2-44.5 or 89.7-101.1
Good 44.6-55.9 or 78.6-89.6
Very Good 56.0-78.5

The Shawangunk Mountains hosts a fire landscape of dwarf pine ridges and pitch pine-oak-heath rocky summits. Most (89%) of the 107 fire records come from the ~58,500 acre northern "firesheds" (Hubbs 1995). Seventy-eight of these fires were assigned at least a categorical size class of <0.25, 0.25–9.9, 10.0–99.9, or ≥100 acres (see figure below). Fires with "exact" size information (58 fires) ranged from <1 acre to over 7,400 acres. Of these, 31 fires burned 100 acres or more, including 14 fires over 500 acres and nine fires over 1,000 acres. Average fire size was 618.7 acres (95% C.I. = 272.0 to 1059.2 from 10,000 bootstrap replicates). Adjusting this fire size to the spatial extent of current Commission-owned lands in the APB, via:

$\sum_{i=1}^{N} (F_i \div 58{,}500 \times 3{,}010) \div N$, where F is the size of the i^{th} fire and N = 58 fires

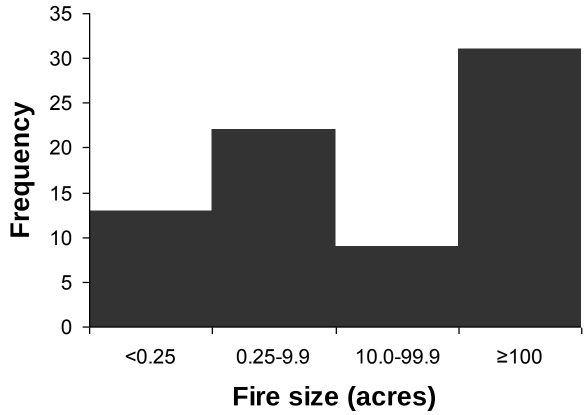
we might expect an average fire size of 31.8 acres and the range 14.0 to 54.5 acres to contain actual fire sizes 95% of the time (i.e., the acceptable or Good range). Using the same logic as for the Long Island ratings, the Very Good range becomes:

$$\frac{54.5+14.0}{2} \pm \frac{54.5-14.0}{4} = 34.3 \pm 10.1 = [24.1\,, 44.4]$$

Thresholds for Good, Fair, and Poor were set by progressively adding and subtracting 10.1 starting from the Very Good limits, ending at <3.9 or >64.6 acre Poor sized fires.

Indicator: *Gunks-based annual mean individual prescribed fire size (acres)*
Poor <3.9 or >64.6
Fair 3.9–13.9 or 54.6–64.6
Good 14.0–24.0 or 44.5–54.5
Very Good 24.1–44.4

Shawangunk Mountains fire size frequency distribution over nearly 150 years (1842–1989). Data from Hubbs (1995).



According to Forman and Boerner (1981), mean annual fire size declined exponentially during 1906–1976 in the New Jersey Pinelands. There was sharp contrast before and after 1940, with 89% of post-1940 averages at or below 10 ha (12.4 acres). Mean annual fire size was at or below 10 ha in over 50% of the 71 year study period. Fires averaged 112.4 ± 20.3 acres annually pre-1940 and 15.8 ± 3.2 acres post-1940. Mean annual fire size was coarsely estimated over the 70 year study period using the plotted data and midpoints of 5 ha increments along the y-axis (2.5, 7.5, 12.5,…97.5) in Fig. 4 of Forman and Boerner (1981). This yielded a mean of 58.6 acres (95% C.I. = 38.5 to 66.7 from 10,000 bootstrap replicates). Using the same logic as before, the Very Good range becomes:

$$\frac{66.7+38.5}{2} \pm \frac{66.7-38.5}{4} = 52.6 \pm 7.1 = [45.5, 59.7]$$

Thresholds for Good, Fair, and Poor were set by progressively adding and subtracting 7.1 starting from the Very Good limits, ending at <31.3 or >73.9 acre Poor sized fires.

Indicator: *New Jersey-based annual mean individual prescribed fire size (acres)*
Poor <31.3 or >73.9
Fair 31.3-38.3 or 66.9-73.9
Good 38.4-45.4 or 59.8-66.8
Very Good 45.5-59.7

Limitations

- ✓ The New Jersey scheme is least reliable because it is not scaled to the APB preserve area.
- ✓ The APB is a much different topographic landscape than the coastal plain Long Island and New Jersey sites and the mountainous Shawangunks. There are further

differences in fuel types, soil characteristics, and weather patterns (e.g., Long Island gets more precipitation and higher mean temperatures than APB).

- ✓ The fire history of each site is incomplete (although probably more complete than for APB) and one should not assume that available records provide fully representative samples. Hubbs (1995) warns that "*many important records of Shawangunk fires have been destroyed or are missing, including most official fire reports*".
- ✓ The measurements ignore possible seasonal (growing vs. dormant) differences in fire size. For example, recorded fire history of the Shawangunks indicates that relatively large fires (100 acres or more) were more common in spring and summer than in fall (Hubbs 1995).
- ✓ Occasional large fires have swept through each of the sites. The current rating scale may undervalue large, infrequent disturbance. Individual fires have burned 400 (twice), 580, 640, and 1,200 acres of APB lands at an average point frequency of 3.3 years (Zaremba et al. 1991). About a third of recorded fires in the Long Island Central Pine Barrens have burned hundreds of acres, with 400+ acre fires occurring at a mean point frequency of 3.0 years during 1968–1989. Managers should beware, however, that a large-scale disturbance covering all or most of a small isolated reserve may erase its "ecological memory" and degrade its spatial resilience (Bengtsson et al. 2003). Moreover, large scales of disturbance (e.g., 400+ acre fires) may not be reproducible in contemporary pine barrens – the urban APB setting, for example, ensures that wildfires are promptly extinguished. The role of management with respect to large and infrequent natural disturbance should be to prepare the site and/or facilitate its recovery process (Dale et al. 1998).
- ✓ Proportion of habitat area burned annually or as the fire season progresses may be a more appropriate way to express the measurement than absolute fire size (Turner et al. 1993, Brockett et al. 2001).
- ✓ Fire size and pattern has important and possibly persistent effects across a landscape (i.e., scale of the APB), but broader-scale gradients may override these effects and ultimately dictate the balance of ecological threats and integrity (Turner et al. 1997).
- ✓ This indicator relies on the historical perspective to provide reference. But must the system be returned to its exact historical range of fire size to provide ecosystem services and support characteristic biota and processes? In this era of rapid change and paucity of reference conditions, we may need to embrace and promote the resilience of novel systems (Seastedt et al. 2008).

Return interval

Rationale: Spatial extent of disturbance together with the interval of disturbance will determine the landscape equilibrium (Turner et al. 1993). High rates of disturbance help increase the longevity of shrublands, savannah, and open woods in the northeastern U.S. (Latham 2003). These early succession systems convert to closed forest and accumulate soil depth (via organic matter accumulation and mineralization) in the absence or reduction of fire (Arabas 2000). Chronic fire is presumed necessary to maintain large expanses of non-forested upland habitat in New England, whereas infrequent fire permits more rapid sprouting and re-colonization of trees (Motzkin et al. 2002). However, a single optimal fire frequency does not exist from the standpoint of maximizing biodiversity, because species are differentially sensitive to fire, even closely related species.

Despite negative, positive, and neutral response to fire among species, a few generalizations are possible for invertebrates: populations are seldom eradicated by single fires (Panzer 2002 and eight references therein), subsurface taxa and life stages are less affected than above-ground stages (Panzer 2002 and five references therein), and post-fire recovery is often rapid (Panzer 2002 and eight references therein). The general expected arthropod response is a strong negative short-term effect, rapid rebounding, and no extirpation (Siemann et al. 1997). Several factors control the insect response, including degree of exposure to lethal temperature and stress, female host selection and oviposition behavior, suitability of post-fire vegetation (niche diversity will be relatively low in recently burned habitat), life history and voltinism, and ability to escape, endure (e.g., hide underground), or colonize (Schowalter 1985, Swengel 2001, Panzer 2002). Wingless species obviously have less ability to escape than winged species. Univoltine species lack extra generations that might otherwise fill the void left by single disturbance events, and thus are probably slower to recover. Fire-sensitive Lepidoptera can be excluded from a shrubland area for several years after it has burned (New 1993, Swengel 1998 and references therein, Wagner et al. 2003). A study of soil invertebrates one year after fire on the Cumberland Plateau reported a 95% total standing stock biomass reduction at the forest floor, with ~60% due to beetle losses (Kalisz & Powell 2000). Univoltine, duff-inhabiting leafhoppers, butterflies, and *Papaipema* moths are considered especially vulnerable to fire-induced extirpation (Panzer 2002 and three references therein).

Fire may strongly depress arthropod abundance, at least over the short term, suggesting that sufficient time between fire events is needed to allow recolonization (Harper et al. 2000). Panzer (2002) looked at insect recovery following spring season prescribed fires in tallgrass and sand prairies and found substantial population declines in 40% of 151 species (representing 33 families and 7 orders) tracked. Proportionately more species of Homoptera (cicadas, leafhoppers, etc) showed negative fire sensitivity than shown by species of Lepidoptera, Orthoptera, Heteroptera, and Coleoptera. Most (68%) populations (163 total) showed mean recovery times of ≤1 year and all populations "recovered" (when post-fire populations were ≥80% of population sizes in unburned controls) within two years during seven seasons of sampling. Panzer (2002) recommended a three-year return interval of rotational, cool-season burning for insect-based grassland management. He warned that annual point burns will limit recruitment of most insect taxa.

Effect of fire return interval (FRT) on birds depends largely on their foraging and nesting habits. For example, ground and low-shrub nesting birds (e.g., northern cardinal, ovenbird) are most likely to experience adverse effects from fire (Artman et al. 2001). Ground foragers are affected because fire consumes bird foods (e.g., acorns, insects) and creates a hotter, drier, and generally unfavorable microclimate for ground-dwelling arthropods (Burke & Nol 1998, Harper et al. 2000, Panzer 2002). Surface fires may affect ground- and low-shrub nesters yet benefit aerial foragers by reducing shrub and sapling density and increasing prey visibility (Artman et al. 2001 and seven references therein). Numbers of ovenbird and black-and-white warbler, both present in the APB, did not recover to pre-burn levels two years after ignition of a pine-grassland community (Wilson et al. 1995). Early succession habitat in the northeastern United States may require a 10–15 year disturbance cycle to maintain shrubland bird assemblages (DeGraaf & Yamasaki 2003).

A general fire frequency would be difficult to define at broad and noisy spatial scales (e.g., Cardille et al. 2001), but independent estimates of historical FRT in pine barrens of the northeastern United States are similar (Forman & Boerner 1981, Windisch 1999, Jordan et al. 2003, Rice et al. 2004). Maintenance of shrublands and shrub savannah in the Long Island Central Pine Barrens probably requires 5–40 year returns of top-killing, high intensity surface or crown fire (Jordan et al. 2003). Returns of less than five years may limit recruitment of scrub oaks, and returns longer than 40 years are less likely to exclude tree oaks and other non-native pine barrens vegetation (e.g., Table 2 in Jordan et al. 2003). In the New Jersey Pinelands, pine-oak forest dominated by oak saplings may develop into oak-pine forest in about 40 years following fire (Forman & Boerner 1981).

Recommendations on FRT for the APB are fairly consistent. An unpublished report by The Nature Conservancy (as cited in Givnish et al. 1988) estimated a historical FRT of 13.9 years. Zaremba et al. (1991) estimated that 2–15 fires swept through the APB each year, and suggested a mean FRT of ~10 years for APB pine barrens maintenance. (Note: the report admits to uncertain fire boundaries and lists numerous small brush and grass fires). Surveys throughout the APB study area in 1980 found that scrub oaks dominated only in sites burnt within the last 20 years (mean 9.4 ± 6.1) whereas stands dominated by red and white oaks had not burned in more than 20 years (Milne 1985). Burning at greater than 20 year intervals may allow black locust, which spreads vigorously through root sprouting, to overtop scrub oaks and create closed-canopy forest (Malcolm et al. 2008). Coring of black locust in parts of the landscape in 2000 revealed last burn times of 15–34 years before present (Rice et al. 2004). Burning pine barrens at 6 to 18 year intervals should maintain “prairie openings” that benefit open-habitat species like the Karner blue butterfly (Givnish et al. 1988).

The fire return interval attribute has two parts, a successional (temporal) component and a spatial component. Pine barrens habitat in the APB grades from relatively open barrens and thicket (scrub oak dominated) to relatively closed-canopy forest (pitch pine dominated). The landscape must be strategically managed to provide a range of complementary habitat for a range of species. For example, a mosaic of interconnected patches buffered by later successional stages is an important feature of suitable Lepidoptera habitat in temperate human-dominated landscapes (Smallidge & Leopold 1997, Grand & Mello 2004). Thus the relative amounts of seral stages and not just the presence of each becomes an important detail.

Habitat longevity is a critical component of habitat suitability in ephemeral systems like shrublands (Latham 2003). The naturally shifting mosaic and contagion of early successional habitat makes it difficult and imprudent to manage for a temporally consistent proportion of cover types, even if the APB has burned frequently and regularly over the last several hundred years (Milne 1985). The successional component is the point-fire frequency cycle needed to maintain areas dominated by pitch pine (~20–40 yr FRT) and areas dominated by scrub oak (~3–20 yr FRT). Patches should be burned at varying times to produce a range of ages or seral stages that will promote a more diverse species mix (Kalisz & Powell 2000, Parr & Andersen 2006). The majority of species in fire-dependent habitats, especially birds and insects, have rapid post-fire recovery times and thus quick fire returns should not be limiting (Anderson et al. 1989, Herkert 1994, Harper et al. 2000, Panzer 2002). The upper limits of 20 and 40 years should favor relatively fire-negative or slow recovering species while still excluding undesirable shrubland vegetation like tree oaks.

The second part of the attribute measurement is to define the spatial distribution of pine barrens habitat experiencing the two fire return intervals. From a historical standpoint, Dettmers (2003) estimates that 10–15% of the northeastern U.S. land base should be managed as early successional habitat to maintain minimal populations of shrubland bird assemblages. This percentage scaled down to the APB landscape would be the equivalent of restoring or maintaining about 1,200 to 1,850 acres of pine barrens in the preserve study area or about 170 to 250 acres of pine barrens in the protected land base. Data from southeastern Massachusetts pine barrens suggest that a majority of barrens habitat should be in early succession or open canopy stages (shrub barrens or thicket instead of forest) for the benefit of breeding birds and rare moths (Grand & Cushman 2003, Grand & Mello 2004). Maintenance of early succession with low tree and shrub cover will benefit the frosted elfin and Karner blue (Albanese et al. 2007, Grundel & Pavlovic 2007), but too little canopy may reduce larval populations (Grundel et al. 1998, Lane & Andow 2003, Albanese et al. 2008).

Assimilating the information above, a desirable (Good) habitat ratio might be 75% open barrens and thicket and 25% pitch pine forest across the landscape at any given time; shrubland landscapes probably should not be managed for complete open barrens (Motzkin et al. 1999). Adult frosted elfin may reach greatest densities with tree cover <29% (Albanese et al. 2007), and a similar canopy threshold was identified for the Karner blue (Grundel et al. 1998). The recent studies of biodiversity response to scale and structure in barrens of Massachusetts (Grand & Cushman 2003, Grand & Mello 2004) suggest that an even ratio of open and closed habitat would not maximize biodiversity, thus a unity ratio (1:1) seems unviable (Fair). A reasonable Poor rating might then be a majority (75%) of closed habitat, or the opposite of the Good rating.

Indicator: *Areal fraction with scrub oak-maintaining FRT (3–20 yr) vs. pitch pine-maintaining FRT (20–40 yr). The measured cover should be rounded to the nearest percentage*
Poor 25% shrub, 75% tree
Fair 50% shrub, 50% tree
Good 75% shrub, 25% tree
Very Good NA

Limitations

- ✓ Site-specific conditions like soil nitrogen levels and invasive species, and broader patterns of fragmentation and climate change, may confound the perception and reality of an ideal fire return interval.

- ✓ The decision for FRT depends partly on whether management is aimed at maintaining current successional states (maintenance fire) or forcing state-transitions (restoration fire). Longer return intervals are allowed for maintenance fires whereas shorter intervals are needed to force state-transitions (Jordan et al. 2003). The proposed FRT rating scheme is geared towards maintenance. However, a major goal in the APB is to initiate restoration in areas that have never burned (to our knowledge) or have not burned in many years, thus a full-scale maintenance regime is not yet the focus.
- ✓ Fire cycle and therefore fire suppression is only part of the equation for fire-dependent taxa. For example, an unburned matrix with heavily stressed or recently expired trees provides important egg-laying habitat and colonization sources for pyrophilous saproxylic taxa. With shortening fire cycles fewer trees senesce before being killed by fire, limiting snag recruitment for wood-feeding species (Saint-Germain et al. 2008).
- ✓ It is short-sighted to credit FRT as the sole mechanism in driving pine barrens succession, because transitions will depend on other factors like proximity to seed sources and amounts of exposed mineral soil and rainfall (Jordan et al. 2003).
- ✓ A 20–40 yr FRT may not be frequent enough to exclude black locust (Malcolm et al. 2008).
- ✓ It is not clear what role frequent, cool, low intensity, patchy fires may have on maintenance of pitch pine-scrub oak communities.

Seasonality

Rationale: The APB fire management program has traditionally focused on growing season burns (APBPC 2002, Gifford et al. 2006). Summer fire may reduce the spread of invasive species (Gebauer et al. 1996) and facilitate recruitment of pitch pine, which is shade-intolerant and inhibited by thick litter (Motzkin et al. 1999). In grasslands of New England, spring burns may be less effective than summer burns and mowing at slowing woody succession, increasing native species richness, and promoting rare species (Dunwiddie 1998).

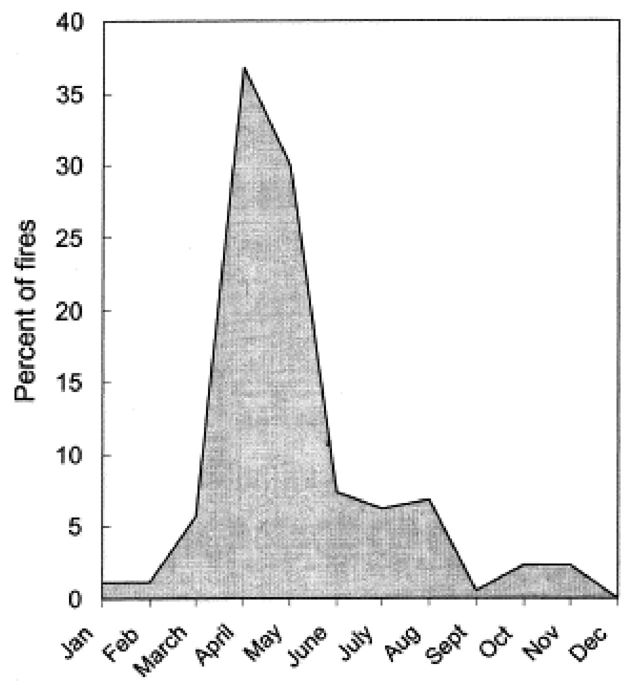
However, climatic conditions are most favorable for pine barrens fires in the early spring (Forman 1979), not to mention that burning regulations limit fire usage during dry periods of summer. Wildfires in the Long Island Central Pine Barrens have occurred mostly during the dormant season over the past 70 years, with over 65% of these occurring in April and May alone (see figure below). Historically, lightning fires lasted from late spring through summer in the APB (Benton 1976), with the greatest number occurring in mid to late April (usually fuels are well cured at this time) and from October to November after leaf fall (Zaremba et al. 1991). Native Americans may have burned more frequently in the fall than any other time of year in the northeastern United States (Russell 1983).

Given these contradictions and assuming that wildfires burned randomly over time, a mixture of growing and dormant season burns may work best. Of the 98 historical Shawangunk fires with known season of occurrence, 38% occurred in spring (March-April-May), 34% occurred in summer (June-July-August), and 27% occurred in fall

(September-October-November) (Hubbs 1995). These records suggest a roughly even historical distribution of dormant season and growing season fire. As such, viable prescribed fire seasonality (Good rating) for APB pine barrens might be a 50:50 split of growing and dormant season burns. The remaining seasonality thresholds were assigned as evenly spaced departures from the desired distribution, using a coarse level of precision (i.e., rounding to the nearest 25%) given the uncertainty.

Indicator: *Annual distribution of growing to dormant season burns (round to the nearest ratio; e.g., 6 growing season burns and 4 dormant season burns is closer to a 50:50 split than any other option)*
Poor 100:0 or 0:100 percent split
Fair 75:25 or 25:75 percent split
Good 50:50 percent split
Very Good NA

Seasonality of fire. Data are for 177 fires for which dates are available, that occurred between 1938 and 1995 in the Central Pine Barrens, Long Island, New York. From Jordan et al. (2003); used with permission.



SUMMARY TABLE

	Key ecological attribute	Indicator	Ratings			
			Poor	**Fair**	**Good**	**Very Good**
Prescribed Fire Regime	Refugia	seasonal amount (%) of spared (unburned) habitat	<25 or >90	25-50 or 80-90	50-60 or 70-80	60-70
	Individual fire size	Long Island-based annual mean (acres)	<33.2 or >101.1	44.5-33.2 or 89.7-101.1	55.9-44.6 or 78.6-89.6	56.0-78.5
		Gunks-based annual mean (acres)	<3.9 or >64.6	13.9-3.9 or 54.6-64.6	24.0-14.0 or 44.5-54.5	24.1-44.4
		New Jersey-based annual mean (acres)	<31.3 or >73.9	38.3-31.3 or 66.9-73.9	45.4-38.4 or 59.8-66.8	45.5-59.7
	Return interval (FRT)	Areal fraction of scrub oak-maintaining FRT (3–20 yr) versus pitch pine-maintaining FRT (20–40 yr)	25% shrub, 75% tree	50% shrub, 50% tree	75% shrub, 25% tree	NA
	Seasonality	annual distribution (% ratio) of growing to dormant season burns	100:0 or 0:100	75:25 or 25:75	50:50	NA

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V. Biotic Patterns

- ✓ Biotic indicators combine the direct, secondary, and cumulative threats, physicochemical stress, and living interactions of the system, and often provide early warning signals (Noss 1990, Kavanaugh & Stanton 2005).

- ✓ Contradictory evidence for conservation umbrella potential (Andelman & Fagan 2000, Fleishman et al. 2001, Caro 2003, Roberge & Angelstam 2004, Rondinini & Boitani 2006, Sergio et al. 2006, Bried et al. 2007) underscores the importance of relying on multiple indicator species. Recent studies have exposed the value of species identity (as opposed to just richness), complementarity, and multi-taxa concepts in conservation (e.g., Vessby et al. 2002, Su et al. 2004, Anand et al. 2005, Fleishman et al. 2006; but see Oliver et al. 1998), including research in a northeastern U.S. pine barrens (Grand et al. 2004). Although a species with strong habitat specificity should be highly vulnerable to modification of its habitat, single species can not span the range of ecological states found in species assemblages (McGeoch et al. 2002, Nicholson & Possingham 2006).

- ✓ Despite obvious limitations of single species indicators, changes in species occupancy or abundance will sometimes coincide with changes in broader taxonomic patterns, making single species reliable surrogates (e.g., Manley et al. 2006). Moreover, rare species may "slip through the pores" of the wider net cast by community and multi-taxa conservation (Lawler et al. 2003). Effective single-species indicators for monitoring will most likely be those that are area-limited, dispersal-limited, resource-limited, process-limited, keystones, narrow endemics, and/or flagships (Landres et al. 1988, Lambeck 1997, Noss 1999).

- ✓ Single species attributes (cover of pitch pine and scrub oak, invasive plant impact, reduction of priority invasives) and multi-species attributes (floristic tolerance of human activity, characteristic rare Lepidoptera, shrubland birds) are built into this part of the pine barrens viability assessment.

Cover of pitch pine and scrub oaks

Rationale: It makes sense to monitor trends in the distribution and abundance of the essential plant species in communities targeted as conservation priorities (Landsberg & Crowley 2004). The fire return interval attribute is linked to the spatial and successional juxtaposition of pitch pine and scrub oak dominated communities, but it falls short of ranking the desirable amounts of these species where they occur. In other words, the indicator focuses on the relative amounts of pitch pine versus scrub oak but ignores their absolute cover.

Many animals are differentially sensitive to amounts of tree versus shrub cover. For example, in southeastern Massachusetts pine barrens, densities of adult frosted elfin

(*Callophrys irus*), a Species of Greatest Conservation Need present in the APB, were greatest when tree cover was <29% and declined when shrub cover exceeded 16% (Albanese et al. 2007, 2008). Approximately 30% of rare Lepidoptera species with obligate association to pine barrens rely on scrub oak as host plant (e.g., *Acronicta albarufa*) or oviposition site (e.g., *Hemileuca maia*), and about a dozen species feed on pitch pine (Wagner et al. 2003). Abundance of whip-poor-wills and two moth species, Gerhard‟s underwing (*Catocala herodias gerhardi*) and Melsheimer‟s sack-bearer (*Cicinnus melsheimeri*), was positively related to the amount (% of landscape) of scrub oak frost pockets at 300–600 m radii in a southeastern Massachusetts pine barrens (Grand et al. 2004).

Pine barrens shrubland habitat in the northeastern U.S. tends to contain less than 60% but greater than 10 to 25% tree cover (Edinger et al. 2002, Jordan et al. 2003). As such, a benchmark of >60% pitch pine cover seems reasonable to establish an area as pitch pine-scrub oak *forest*. Thinning of dense pitch pine stands and preventing canopy closure (>90% cover) will likely benefit the majority of barrens-dependent moths and birds (Grand & Cushman 2003, Grand & Mello 2004), thus an upper viability limit of 90% pitch pine cover makes sense even in forest. In loblolly-shortleaf pine stands of the coastal piedmont, probability of occurrence of shrubland birds like common yellowthroat, eastern towhee, and indigo bunting, all found in the APB, declined rapidly with increasing canopy cover, whereas mature-forest birds like pine warbler showed the opposite trend (Caterbury et al. 2000); this study suggests that moderate levels of canopy cover (e.g., 40–60%) may benefit the most pine barrens bird species. For scrub oaks in the APB, the working hypothesis is that approximately one-third cover of *Quercus ilicifolia* and *Q. prinoides* may be ideal (Very Good) across scrub oak-dominated pine barrens, with increasing thicket less desirable. Areas with too little scrub oak cover (say <20%) may attract parasitoids of barrens buckmoth larvae in the APB and elsewhere (Selfridge et al. 2007; D. Parry, State University of New York College of Environmental Science and Forestry, unpublished data), providing a basis for the Poor rating.

Indicator: *cover of pitch pine across pitch pine-scrub oak forest*
Poor <20 or >90
Fair 20–40
Good 40–60 or 75–90
Very Good 60–75

Indicator: *cover of scrub oak across pitch pine-scrub oak barrens and thickets*
Poor <20 or >75
Fair 50–75
Good 35–50
Very Good 20–35

Limitations

- ✓ Historical sources provide little information about the relative importance of pitch pine vs. white pine and scrub oaks vs. tree oaks in xeric outwash, pine plains, and pitch pine-scrub oak communities, making it difficult to define reference conditions of stand composition and structure (Motzkin et al. 1999).

Floristic tolerance of human activity

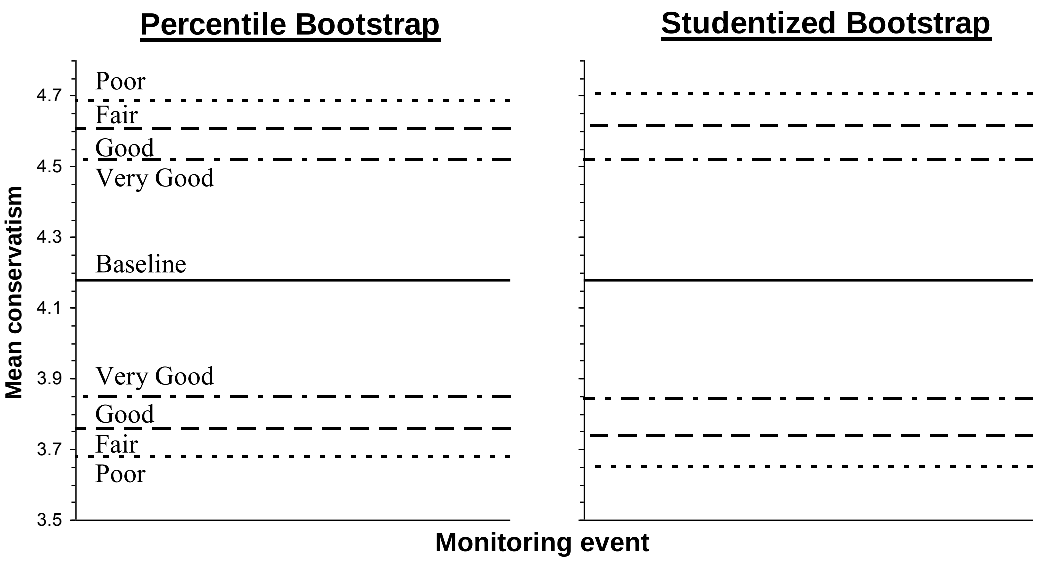
Rationale: Conservatism measures the propensity for plant species to occur in human-dominated habitat. Highly conservative species show a high degree of fidelity to a narrow range of habitats and human disturbance whereas non-conservative species (e.g., exotics, ruderals) show a high degree of ecological tolerance and tend to occupy a variety of plant communities. This concept is widely used for terrestrial areas monitoring and evaluation in the United States (Herman et al. 1997, Panzer & Schwartz 1998, Francis et al. 2000, Allison 2002, Poling et al. 2003, Rothrock & Homoya 2005, Bowles & Jones 2006, Jog et al. 2006, Spyreas & Matthews 2006, Taft et al. 2006).

At present New York and New England do not have lists of conservatism coefficients, so the current rating scale is based on New Jersey coefficients (BHWP 2006). Cumulative species lists were compiled from 1991 and 1993 surveys of 21 permanent plots scattered throughout the APB pine barrens habitat (Gebauer et al. 1996). A total of 112 vascular species were observed in 1991 and 98 vascular species were observed in 1993; New Jersey conservatism coefficients were available for all species. A control chart (see Morrison 2008) was established using the 1991 data as a baseline (see figure below). Mean conservatism was used for the centerline with action thresholds set at the 80, 90, and 95% bootstrapped confidence limits. Two bootstrap methods were run, percentile and studentized (see Dixon 2001).

Results were very similar between years and bootstrap methods, so the 1991 percentile bootstrap was used for indicator ratings. The monitoring plan is to periodically resample the same 21 plots as close in time as possible, compute the average conservatism, and plot the points in the figure shown below. For example, the 1993 resurvey of all plots yielded a mean conservatism of 4.48, which puts the target barely in the Very Good range.

Indicator: *bootstrapped confidence intervals for mean conservatism of total species detected*
Poor <3.68 or >4.69
Fair 3.68–3.76 or 4.61–4.69
Good 3.77–3.85 or 4.52–4.60
Very Good 3.86–4.51

Indicator thresholds set at the 2.5th and 97.5th percentiles (Very Good-Good), 5th and 95th (Good-Fair), and 10th and 90th percentiles (Fair-Poor). The baseline represents the average conservatism from cumulative plant species observed in 1991 across 21 permanent plots (50 × 20 m) in pitch pine-scrub oak barrens, thicket, and forest.



Limitations

- ✓ The entire indicator range (Poor to Very Good) covers a narrow portion of the 0 to 10 conservatism scale. Typically *C* values of 3 or 4 indicate species bordering on high to intermediate levels of ecological tolerance, and/or species that do not typify advanced successional communities (Andreas et al. 2004, BHWP 2006). Therefore, the current rating categories may have similar ecological meaning. Low (*C* = 0 to 3), moderate (*C* = 4 to 6), and high (*C* = 7 to 10) categories and not the integers or real numbers may be the more informative resolution.
- ✓ The measure assumes there are upper and lower limits to historical conservatism levels, which means that managers should not aim in one direction only (i.e., may have to raise or lower aggregate conservatism). The desired state of pine barrens is an early rather than advanced successional community. Naturally dry, acidic, and poor soils may select for predominantly weedy and invasive species. If true, the desired state of barrens could be one of relatively low aggregate conservatism, or the reverse idea of the traditional conservatism scale (originally built for prairies in the Chicago Wilderness region; Swink & Wilhelm 1979). Alternatively, barrens succession may *not* favor weedy and invasive species, due to strong positive feedbacks in which dominant native species contribute to environmental changes that tend to favor their own persistence (Latham 2003). The issue is further confused by the fact that pine barrens vegetation is dependent on natural or simulated natural disturbance and thus may have inherent resilience to undesirable human disturbance. Given the uncertainty, a two-tailed hypothesis and thus the two-way thresholds seem reasonable.
- ✓ The indicator sensitivity is based strictly on species turnover, not abundance. Overall conservatism increased when species‟ conservatism values were weighted

by abundances (total point-intercepts). In 1991, straight-average conservatism was 4.18 whereas weighted-average conservatism was 4.97, and in 1993 straight-average conservatism was 4.48 compared to weighted-average conservatism of 5.04. The six most abundant (common) species accounted for 51% of total point-intercepts and had relatively high conservatism (*C*) scores: *Quercus ilicifolia* (*C* = 7), *Vaccinium pallidum* (*C* = 7), *Quercus prinoides* (*C* = 8), *Carex pensylvanica* (*C* = 9), *Pinus rigida* (*C* = 6), *Gaylussacia baccata* (*C* = 8). The next six most abundant species had relatively low conservatism scores and accounted for only 18% of total point-intercepts: *Pteridium aquilinum* (*C* = 2), *Lysimachia quadrifolia* (*C* = 3), *Prunus serotina* (*C* = 1), *Rubus allegheniensis* (*C* = 3), *Robinia pseudoacacia* (*C* = 0), *Populus tremuloides* (*C* = 2).

- ✓ Some New Jersey conservatism values may be different from New York. A New York conservatism list is needed.

Invasive plant impact ("invasiveness")

Rationale: Members of the New York Invasive Plant Council recently completed an invasiveness ranking protocol for plants alien to New York. The protocol borrows heavily from the criteria developed by Alaska Natural Heritage Program (Carlson et al. 2008), which has elements of the National Park Service and Natural Heritage Network ranking systems (Hiebert & Stubbendieck 1993, Randall et al. 2008). The system combines 22 criteria of structural and functional impact (40% weight), biological traits like reproductive mode and competitive ability (25%), ecological amplitude and distribution (25%), and feasibility of control (10%). The final scoring rank (*IScore*) ranges from 0 (no current or potential impact) to 100 (maximum current or potential impact).

The „Poor" ranking assumes that even one "high-threat" species (*sensu* Parkes et al. 2003) can severely hurt pine barrens ecological integrity. Aspens and black locust are examples of high-threat species in the APB. The proposed indicator thresholds are arbitrary (quartiles) and not meant to reflect shifts in ecosystem state, unlike true thresholds in ecology and adaptive management (Groffman et al. 2006). "Uncontrolled" in the measurement definition allows room for expert judgment by managers and scientists – while some invasive species may never be eradicated, they may be reduced to perceived acceptable levels.

Indicator: *maximum invasiveness ranking ($IScore_{max}$) of uncontrolled exotic/native species*
Poor ≥75
Fair 50-74
Good 25-49
Very Good <25

Indicator: *current weed cover*
The previous indicator is weighted towards the potential for species to invade, with less emphasis on existing distribution and abundance levels. In describing a „habitat hectares"

approach to remnant vegetation assessment, Parkes et al. (2003) recommended a component scored on four levels of total weed cover (<5; 5–25; 25–50; >50%) and three levels of cover by high-threat weeds (none; ≤50%; >50%). Trusting in the generality and basis for this measure, and given the 12 total combinations of weed cover thresholds, the most reasonable fit to the four-part rating system might look like:

Weed cover	% of weed cover due to high-threat species		
	None	**≤50%**	**>50%**
>50%	Fair	Poor	Poor
25-50%	Good	Fair	Poor
5-25%	Very Good	Good	Fair
<5%	Very Good	Very Good	Good

„High-threat" species defined by *IScore* ≥ 75; adapted from Parkes et al. (2003)

Limitations

- ✓ An obvious limitation is that cover sampling requires lots of effort and resources, and ideally the abundance of invasive species should be sampled across the APB. However, existing permanent plots (Gebauer et al. 1996) should enable representative floristic assessment of the target area (see box below).

Representativeness of permanent plots sampling for estimating floristic quality attributes

To estimate whether permanent plot sampling was representative of the APB landscape, the full species checklist in the 1991 baseline survey (see Gebauer et al. 1996) was measured against the authoritative list of APB plant species in Barnes (2003). Statistical differences in floristic quality variables between these lists were tested using means (*t* test) or proportions (*z* test).

Analysis of sampling completeness suggests that permanent plots provide a representative floristic quality estimate of the APB landscape. In the 1991 baseline survey 231 plant species were recorded. Although this checklist included only 31% of the confirmed plant species in the APB (based on Barnes 2003), there was no evidence of difference in mean conservatism between the Barnes" checklist and the baseline plot survey (unequal variance t-test, $t = 0.465$, $p = 0.642$). Barnes listed 3.5× more total species than observed in plots, but included a similar proportion ($z = 0.821$, $p = 0.471$) of conservative species (defined as having C of 8, 9, or 10). Plots may, however, underestimate the number of exotic species ($z = 3.389$, $p = 0.001$).

Reduction of priority invasive vegetation

Rationale: Biological invasion is a leading cause of ecosystem dysfunction and biodiversity loss on the global stage. Priority invasive species in the APB are native aspens (*Populus grandidentata*, *P. tremuloides*) and exotic black locust (*Robinia pseudo-acacia*) (APBPC 2002). Nitrogen-fixing black locust ranks as the second most abundant deciduous tree worldwide and is notorious for altering nutrient cycles in grassland and barrens ecosystems (Rice et al. 2004). It not only enriches naturally poor soils, but also builds excessive litterfall and closed canopies that compete with native plant growth and recruitment (Rice et al. 2004, Malcolm et al. 2008). Aspens take advantage of frost tolerance and fire suppression and usurp large areas of the APB landscape through rapid clonal establishment (Milne 1985). Shrubland birds like prairie warbler, field sparrow, and eastern towhee prefer uninvaded areas of the APB landscape (Beachy & Robinson 2008, Gifford et al. *in review*).

Initial thresholds were set by comparing recent land cover classification data against reduction objectives in the APB management plan (APBPC 2002). Reduction objectives evolved from over 15 years of experience by the Commission and its partners with managing the preserve. The management plan (pages 41–42) calls for at least 50% reduction of black locust and 90% reduction of aspen across the preserve by 2012. As of May 2003, locust had colonized roughly 745 acres and aspens roughly 253 acres of Commission-owned lands and agreements (map analysis by B. Kinal, former APB Preserve Ecologist). The management plan and land cover analysis were completed at about the same time, thus the estimated coverage of aspens and locust in 2003 may be used as benchmarks for evaluating progress towards the reduction objectives. The reduction thresholds were set using starting values of 745 acres (for locust) and 253 acres (for aspen). The 10-acre threshold for aspen assumes that eradication is not desirable because aspens are native to pine-oak barrens.

Indicator: *preserve-wide cover of black locust remaining*
Poor ≥559 acres
Fair 558–373 acres
Good ≤372 acres (≥50% reduction)
Very Good no locust

Indicator: *preserve-wide cover of aspen remaining*
Poor ≥139 or <10 acres
Fair 139–26 acres
Good 26–10 acres (≥90% reduction)
Very Good NA

Limitations

✓ The measurement scheme applies preserve-wide rather than explicitly to pine barrens. Managers should adjust the ratings to account for the percent of reduction that occurs in pitch pine-scrub oak remnants.

Characteristic rare Lepidoptera

Rationale: Arthropods serve diverse taxonomic and functional roles, occupy basal or mid-level consumer positions in trophic webs, and show a wide range of body sizes and vagilities (Kremen et al. 1993). The number of species and sheer abundance of invertebrates is paramount to biodiversity patterning and ecosystem function at all spatial scales.

Some evidence suggests that the more commonly applied vertebrate- or plant-based conservation schemes are inadequate for invertebrate protection (Oliver et al. 1998, Rubinoff 2001, Axmacher et al. 2004). Panzer & Schwartz (1998) cited 12 primary publications that criticized plant- and vertebrate-based conservation schemes for invertebrate protection. This comes as no surprise given that spatial patterning of invertebrates is confined to smaller scales than that of vertebrates (Mac Nally et al. 2004). Grand et al. (2004) found little overlap between bird and moth rarity hotspots in a southeastern Massachusetts" pine barrens. Although the hotspots afforded high levels of cross-taxon species representation, the authors warned that large numbers of species still might be missed by single-taxon conservation schemes. And even when cross-taxon schemes capture a large fraction of the community, rare and at-risk species may still be overlooked (Lawler et al. 2003).

In many cases passive surveys required for arthropods are easier and less costly than vertebrate sampling (Underwood & Fisher 2006, Rohr et al. 2007). However, complete enumeration of speciose taxa like insects is generally impractical (Kremen et al. 1993). It may prove futile, for example, to monitor the hyper-diverse moth fauna of the APB (550 noctuid species alone; Barnes 2003). Monitoring programs should choose easily sampled (relatively speaking) invertebrates known or hypothesized to indicate abundance and distributions of other invertebrates (Thomas 2005). A subset of a chosen invertebrate indicator assemblage may be used as a conservation surrogate for the remaining species (Fleishman et al. 2000, Bried et al. 2007).

Lepidoptera assemblages can serve as powerful indicators of disturbance (Kitching et al. 2000), and compared to other phytophagous insects with similar levels of specialization (e.g., thrips, true bugs, leaf beetles) in pine barrens, they are relatively easy to work with. Subsets of species might be monitored in lieu of the total assemblage (Swengel & Swengel 1997). Several dozen rare Lepidoptera species show obligate pine barrens association (Wagner et al. 2003), thus simple diversity measures (composition and richness) may provide a strong signal of pine barrens degradation. The presence of rare pine barrens Lepidoptera provides an integrated picture of nutriment supplies, canopy cover, edge contrast, core area, patch density and shape, and frost pockets (Wagner et al. 2003, Grand & Mello 2004). Although species behave differently according to environmental gradient and scale, effective management of some species may benefit sympatric Lepidoptera and other rare insects that thrive in shrublands (Swengel & Swengel 1997, Albanese et al. 2007).

Wagner et al. (2003) provide a comprehensive list of rare shrubland Lepidoptera in southern New England and eastern New York. This list includes seven of the ten rare moth species studied in a Massachusetts coastal plain pine barrens (Grand & Mello 2004) and nearly all of the lepidopteran Species of Greatest Conservation Need documented (at least historically) in the APB (Givnish et al. 1988). The APB checklist is a subset of the Wagner et al. (2003) checklist. Preliminary indicator thresholds below are based on distributional data of rare, obligate Lepidoptera across 11 sites in the northeast, including two in Maine, two in New Hampshire, two in Massachusetts, two in Connecticut, two in New York (including the APB), and one in Rhode Island (Table 3 in Givnish et al. 1988). According to Givnish et al. (1988), species-area relationships (Connor & McCoy 1979) were strongly linear using current (83% of richness variation explained) and historical (79% explained) area estimates.

An improved species-area equation (regression parameters) using the same data in Givnish et al. (1988) was built by shuffling the data 10,000 times with replacement. The regression slope and intercept were computed at each iteration using the built-in regression function in Microsoft Excel. Using ln-transformed number of species present as of ca. 1988 and ln-transformed historic area, mean slope = 0.331 (95% CI limits of 0.209 and 0.416) and mean intercept = -0.642 (95% CI limits of -1.379 and 0.526). Using ln-transformed number of species present and collected previously but believed absent (possibly extirpated), and ln-transformed historic area, mean slope = 0.336 (95% CI limits of 0.195 and 0.486) and mean intercept = -0.509 (95% CI limits of -1.790 and 0.627). The former set of parameter estimates were chosen for the equation because: (1) this model explained 10% more of the variation in species number, and (2) "presences" are unambiguous (assuming species are correctly identified) whereas "absences" are prone to detection error (MacKenzie et al. 2006). The thresholds for the „Habitat amount‟ (A) attribute in II. Size & Extent (i.e., Poor-Fair at 1000; Fair-Good at 2000; Good-Very Good at 8,500) were used to estimate the equilibrium species number (S) at each threshold with the relationship [$\ln S = 0.331(\ln A) - 0.642$]. The solutions are taken as preliminary indicator ratings.

Indicator: *number of rare characteristic species observed in the target*
Poor <5
Fair 5–7
Good 8–11
Very Good >11

Limitations

- ✓ "Restored" barrens may lack the invertebrate fauna of historically unaltered habitat (Kirby 2001), confusing the use of "characteristic" species as a restoration indicator. Moreover, the time lag between restoration treatment and presence of characteristic species may falsely indicate less than desirable conditions when they actually occur.
- ✓ The analysis follows the very restrictive assumption that species number and habitat area are in equilibrium across the region.
- ✓ The Givnish et al. (1988) list appears incomplete or outdated compared to Wagner et al. (2003).
- ✓ Species recorded as present contribute to species richness, but the amount, configuration, or suitability of habitat in the landscape may be below their extinction threshold (i.e., probability of persistence <1) (see Radford et al. 2005).
- ✓ Thresholds for species richness are often weakened by contrasting responses of individual species (Lindenmayer et al. 2008).

Shrubland birds

Rationale: Birds are well studied, relatively easy to sample, and useful for monitoring ecological change and conditions across a wide range of ecosystems (e.g., Beintema 1983, Burger et al. 1994, Keddy & Drummond 1996, Bradford et al. 1998, Canterbury et al. 2000, Marzluff & Ewing 2001, Bryce et al. 2002, Diamond & Devlin 2003, Hausner et al. 2003, Mac Nally et al. 2004, Tankersley 2004). Birds also are sensitive to threats like fire repression, fragmentation, and urbanization (e.g., Kerlinger & Doremus 1981, Robbins et al. 1989, Hunter et al. 2001, Dettmers 2003, Lorimer & White 2003, Manley et al. 2006).

Different bird species and functional groups will respond differently to threat situations and habitat conditions (e.g., Odell & Knight 2001, Grand & Cushman 2003, Manley et al. 2006). Few disturbance-dependent bird species are restricted to one habitat type (Hunter et al. 2001), but species like whip-poor-will, prairie warbler, eastern towhee, eastern wood-pewee, brown thrasher, northern bobwhite, black-billed cuckoo, and great crested flycatcher are commonly associated with shrublands and thus could make useful indicators of early successional health and biodiversity (DeGraaf & Yamasaki 2001, Grand & Cushman 2003). Analyses of breeding bird data from the APB revealed clear indicator species for scrub oak barrens/thickets (e.g., prairie warbler, eastern towhee) and pitch pine forest (e.g., black-capped chickadee, chipping sparrow) (Beachy & Robinson 2008; Gifford et al. 2009).

Indicator: *Bird-community index that contrasts the number of disturbance-sensitive (mature forest,* MF*) species against the number of disturbance-dependent species (shrubland,* S*) via the expression* $\ln(S + 1) - \ln(MF + 1)$ *(Canterbury et al. 2000)*
Poor below -0.8
Fair -0.8 to -0.6
Good -0.6 to neutral
Very Good positive value

A negative value indicates a majority of advanced successional species whereas a positive value indicates a majority of early successional species. A reasonable goal is for the APB to attract early successional (shrubland) species. A detailed bird survey along the APB trail system in 2005 found 24 species typically associated with shrublands and 28 species typically associated with mature forest (Gifford et al. *in review*). Associations were based on a synthesis of life history accounts (Kaufman 1996, Levine 1998, DeGraaf & Yamasaki 2001) by Nathali Neal (Union College, Schenectady, New York). The current (2005) bird-community index value of -0.43 sets the baseline for the rating scheme. The „Good" lower limit allows loss of up to two shrubland species and gain of up to two mature forest species. The „Fair" lower limit allows loss of up to four shrubland species and gain of up to four mature forest species. Any further loss or gain results in a „Poor" rating.

Limitations

- ✓ Density may be more informative than a species checklist, but is probably still less meaningful or robust than fitness indicators (e.g., nesting success) as a measure of habitat quality (van Horne 1983). Some experts argue that monitoring programs using birds should measure the production, survival, and dispersal of individuals to adequately gauge restoration progress, despite the greater difficulty and expense of obtaining such information (Marzluff & Ewing 2001). However, some of this information may be inferred from other attributes, such as inter-patch distances as a surrogate for dispersal likelihood or success. Moreover, bird abundance and fitness data can be less reliable than presence data because of high natural variation in population levels and number of breeding pairs (Keddy & Drummond 1996, Mac Nally et al. 2004).

- ✓ Guild theory has long recognized the potential for extrapolating a stress response of one guild member to the larger guild (Severinghaus 1981). Guilds can be useful for evaluating the collective response of multiple species to changes in resources or ecological conditions that define the guild (Verner 1984, Karr 1991, Block et al. 1995). An improved bird-community integrity index, built from only a species checklist, might combine species richness with tolerance to human disturbance, foraging and dietary guilds, and nesting strategies (see Table 2 in Bryce et al. 2001). Combining these trait-based guilds into a multimetric index may result in greater precision (Karr 2000). This should be a future step in viability indicator development for pine barrens.
- ✓ Efforts to conserve rare species or richness hotspots in landscapes may not be effective in protecting broader vertebrate diversity (Chase et al. 2000).

SUMMARY TABLE

	Key ecological attribute	Indicator	Ratings: Poor	Fair	Good	Very Good
Biotic Patterns	Cover of pitch pine and scrub oaks	cover (%) of pitch pine across PPSOF	<20 or >90	20–40	40–60 or 75–90	60-75
		cover (%) of scrub oaks across PPSOB/T	<20 or >75	50–75	35–50	20–35
	Floristic tolerance of human activity	bootstrapped confidence intervals for mean conservatism of total species detected	<3.68 or >4.69	3.68–3.76 or 4.61–4.69	3.77–3.85 or 4.52–4.60	3.86–4.51
	Invasive plant impact	maximum invasiveness ranking ($IScore_{max}$) of uncontrolled exotic/native species	≥75	74–50	49–25	<25
		current weed cover	matrix format – see narrative			
	Reduction of priority invasive vegetation	acreage of black locust remaining	≥559	558–373	≤372	no locust
		acreage of aspen remaining	≥139 or <10	139–26	25–10	NA
	Characteristic rare Lepidoptera	number of rare characteristic species	<5	5–7	8–11	>11
	Shrubland birds	ln(S + 1) – ln(MF + 1), where S = shrubland species richness and MF = mature forest species richness	below -0.8	-0.8 to -0.6	-0.6 to neutral	positive value

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VI. Other Potential Attributes

Ants

Ants (Hymenoptera: Formicidae) are valuable for conservation monitoring (Brown 1997). They are sensitive to soil properties, vegetation, fire regime, patch area, and fragmentation (Carvalho & Vasconcelos 1999, Golden & Crist 2000, Bestelmeyer & Wiens 2001, Braschler & Baur 2003, Izhaki et al. 2003; but see Dauber et al. 2006, Gibb & Hjältén 2007), and have long been used as indicators of land management practices and long-term ecosystem change (reviews by Andersen & Majer 2004, Underwood & Fisher 2006). Human effects on ants are both disturbance-mediated (i.e., removal of biomass) and more indirect or stress-related via changes in habitat structure, microclimate, and food availability (Andersen & Majer 2004).

A comprehensive reference is available explaining how to sample ground-dwelling ants (Agosti et al. 2000). Low-cost passive techniques like pitfall traps and litter sampling are effective (Underwood & Fisher 2006). Time saver strategies include sorting to morphospecies or functional feeding groups and using lower taxonomic resolution (Beattie & Oliver 1994, Andersen 1995, Oliver & Beattie 1996). For example, Andersen et al. (2002) sorted large ant morphospecies (4 mm length threshold) and retained species-level precision, reduced effort by 90%, and reproduced virtually all of the key information gained by intensive surveys. These authors claimed that most large ant species can be successfully sorted by amateurs, but admitted their protocol would be less effective in cool-temperate zones with relatively low ant diversity and smaller ant species.

Fire severity

In addition to chronic fire, occasional severe fire is likely needed to maintain early succession xeric shrublands in New England (Motzkin et al. 1999, 2002). Periodically burning off excess litterfall biomass (dominated by scrub oaks in APB pine barrens; Rice et al. 2004) will serve to reduce fuel hazards for prescribed burns and wildfires, release nutrients and stimulate plant growth, and create open-space microhabitats for the benefit of numerous arthropods (Arabas 2000, Kirby 2001). Occasional severe fires will also expose mineral soil for seedling establishment, which is important for species like pitch pine (Good & Good 1975, Ledig & Little 1979). At the same time, low litter amounts should limit establishment of undesirable tree oaks.

Use of fire severity as an indicator would require establishing fixed-point sampling areas where litter depth measurements are taken at intervals before and after fire, with severity calculated as percent loss or centimeter reduction in litter depth.

Frost pockets

In pine barrens extreme radiational cooling may occur in sand dune depressions, near the base of slopes, or even on level plains (Motzkin et al. 2002). These "frost pockets" are known to increase shrubland longevity in sites capable of supporting forest (Aizen & Patterson 1995, Latham 2003).

Frost pockets provide important spring feeding habitat for many regionally rare Lepidoptera, providing highly nutritious leaves and extending the time window of leaf availability (Wagner et al. 2003, Grand & Mello 2004). Frost pockets may also inhibit the growth of competitive overstory vegetation (Grand & Mello 2004). The shorter frost-free growing season found in frost pockets causes more frequent dieback of opening leaves, slow growth rate, and a shorter average stem height when compared to other microclimates (Motzkin et al. 2002). This repeated stunting of growth slows the pine barrens to forest successional trajectory.

Herpetofauna

The APB supports a rich diversity of amphibians and reptiles (Stewart & Rossi 1981, Hunsinger 1999); a vast literature has established herpetofauna as valuable biological, environmental, and threat-based indicators.

Mammals

Mammals are popular indicators for ecosystem-based monitoring and adaptive management (Landres et al. 1988). Many mammal species are disadvantaged by human activity (Kavanagh & Stanton 2005). Presence of keystone and large apex carnivores may indicate, among other things, an intact food chain and/or sufficient habitat continuity (Sergio et al. 2006). Many mammalian carnivores require large habitat patches to accommodate wide home ranges and low population densities, and thereby may offer logical minimum area criteria for reserve design (Noss 1999). Indeed, the current „patch size‟ indicator ratings (see II. Size & Extent) incorporate area requirements of large mammals.

Robber flies

Presence/absence over time of four predatory asilids, including *Cyrtopogon lutatius*, *Laphria cinerea*, *Laphria virginica*, and *Proctacanthus rufus*, may help to indicate the general ecological condition of APB pine barrens (McCabe & Weber 1994, Wagner et al. 2003).

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