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Prioritizing land management efforts at a landscape scale: a case study using prescribed fire in Wisconsin

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Abstract. One challenge in the effort to conserve biodiversity is identifying where to prioritize resources for active land management. Cost–benefit analyses have been used successfully as a conservation tool to identify sites that provide the greatest conservation benefit per unit cost. Our goal was to apply cost–benefit analysis to the question of how to prioritize land management efforts, in our case the application of prescribed fire to natural landscapes in Wisconsin, USA. We quantified and mapped frequently burned communities and prioritized management units based on a suite of indices that captured ecological benefits, management effort, and the feasibility of successful long-term management actions. Data for these indices came from LANDFIRE, Wisconsin's Wildlife Action Plan, and a nationwide wildland–urban interface assessment. We found that the majority of frequently burned vegetation types occurred in the southern portion of the state. However, the highest priority areas for applying prescribed fire occurred in the central, northwest, and northeast portion of the state where frequently burned vegetation patches were larger and where identified areas of high biological importance area occurred. Although our focus was on the use of prescribed fire in Wisconsin, our methods can be adapted to prioritize other land management activities. Such prioritization is necessary to achieve the greatest possible benefits from limited funding for land management actions, and our results show that it is feasible at scales that are relevant for land management decisions.

Key words: conservation planning; cost–benefit analysis; land management; LANDFIRE; wildlife action plan; Wisconsin, USA.

INTRODUCTION

A major challenge facing biodiversity conservation is the need to identify where to apply management actions, including restoration efforts. Conservation planning has traditionally focused on identifying new areas on the landscape that, if protected, would help to effectively and efficiently achieve long-term biodiversity conservation goals (e.g., Margules and Pressey 2000, Myers et al. 2000). These methods are now well developed and in use worldwide (e.g., Moilanen et al. 2009). However, the majority of conservation lands require some level of active management or restoration, such as controlling invasive species, planting native vegetation, or maintaining disturbance regimes (Salafsky et al. 2002). These management needs often far exceed the available resources of conservation agencies and organizations (Brooks et al. 2006, Wilson et al. 2006).

When resources are limited, there is a need to prioritize management efforts among projects, sites, species, or ecosystems (e.g., Naidoo et al. 2006, Wilson et al. 2007). One method that can be used to identify priorities is cost–benefit or return on investment analyses (Joseph et al. 2009, Withey et al. 2012). A cost–benefit approach to prioritizing potential conservation projects for action has been used to rank resource allocation among threatened species projects in New Zealand (Joseph et al. 2009), prioritize invasive species management in Australia (Carwardine et al. 2012), and rank ecoregions and countries in Africa where conservation efforts are most likely to be successful (Tear et al. 2014). Strengths of cost–benefit analyses include the inclusion of cost in the model and its high level of transparency, particularly when value judgments are used to define benefits (Game et al. 2013, Tear et al. 2014). Cost–benefit analysis may be valuable for prioritizing land management actions in

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general, and the management of disturbance-dependent ecosystems in particular (Ager et al. 2013), because the model can integrate ecological benefits and non-ecological factors (e.g., cost) that influence land management decisions.

In disturbance-dependent ecosystems that are maintained by wildland fire, the prioritization of management action is typically focused on minimizing the effects of wildfires. Such prioritizations identify regions at high risk for wildfires (e.g., Keane et al. 2010, Thompson et al. 2011) or priority areas for fuel treatments (e.g., prescribed fire, thinning) to minimize damage to homes and resources (e.g., Hessburg et al. 2007). However, the implementation of fuel treatments to minimize wildfire impacts can conflict with regulations, or needs of threatened and endangered species, and be hampered by lack funding (Collins et al. 2010). Our research question was somewhat different though, because we asked whether a cost–benefit approach could be used to prioritize management of frequently burned vegetation types across broad areas with a focus on ecosystem conservation and use of prescribed fires.

Prescribed fire is a land management and restoration tool that has strong ecological benefits along with important non-ecological impacts and limitations. In ecosystems where frequent fire is necessary to maintain species composition and function, land managers can apply prescribed fires to mimic natural processes and meet management goals (Wade and Lundsford 1989). For example, many grassland and savanna ecosystems that are subject to invasion by both native and invasive woody species can be maintained with prescribed fire (Lehman et al. 2014). In these cases, prescribed fires can maintain the conservation values of rare grassland ecosystems (Peterson and Reich 2001) and prevent succession to forest ecosystems (Scholes and Archer 1997).

In natural community types that historically burned frequently, prescribed fires can simulate many of the characteristics of natural fires and help to maintain ecosystem structure and function. Low-intensity prescribed fires open space for plant recruitment in grassland and savanna ecosystems, cycle nutrients, create a mosaic of successional classes on the landscape, and maintain the overall vegetation structure (Scholes and Archer 1997, Hoffmann and Solbrig 2003, Werner and Prior 2013). Prescribed fires can also help to maintain biodiversity, especially via the maintenance of diverse species of herbaceous plants, in grass-dominated ecosystems (Peet and Allard 1993, Platt 1999) and sustain critical habitat for threatened and endangered species (Pendergrass et al. 1999, Scott et al. 2005).

While the ecological benefits of prescribed fires are generally high, prescribed fires can be costly, and the feasibility of applying prescribed fire is strongly affected by a site's proximity to residential areas (e.g., Costanza et al. 2013). The costs associated with prescribed fire include unit preparation and fuel type, staff time and travel, and liability insurance (Hesseln 2000). Calculations of prescribed fire cost have such a broad range (e.g., from US\$3–\$30 per acre [Wood 1988] to US\$10–\$90 per acre [H. Spaul, *unpublished data*]) that assigning a cost can be difficult. Even when resources are available, prescribed fires may not be conducted due to potential risks from smoke, direct damage to property (Dellasala et al. 2004, Knight et al. 2010), or inopportune weather. Although the public often recognizes the ecological need for fire, they can be skeptical of the safety of prescribed fire (Shindler et al. 2009). Indeed, land managers identify the risks inherent with burning near homes and a general public perception of prescribed fire as dangerous as major limitations to its use (Costanza and Moody 2011, Quinn-Davidson and Varner 2011, Costanza et al. 2013). Any approach used to prioritize the application of prescribed fire across the landscape must account for these complexities and challenges that land managers face in applying prescribed fire as a management tool.

Our goal was to prioritize the use of prescribed fire as a land management tool across a large area using an approach with high transparency, understandability, and flexibility for a variety of end users. We had two specific objectives: (1) to quantify and map frequently burned communities, and (2) to prioritize areas for applying prescribed fire based on anticipated ecological benefits, the estimated management effort required to apply fire, and the feasibility of successfully applying fire to sites over the long term. We chose the state of Wisconsin as our study area because of the presence of numerous globally rare grasslands and savannas (Wisconsin DNR 2014) and other natural communities dependent on frequent fire, and because practitioners across multiple natural resource management agencies and conservation organizations were actively requesting help in prioritizing their prescribed fire efforts.

METHODS

Study area

Our study area was the state of Wisconsin, USA, covering approximately 140000 km^2 in the north-central USA. The state is biologically diverse and is located at the confluence of the northern forest, eastern temperate forest, and Great Plains ecoregions (Commission for Environmental Cooperation 1997). The state is bisected by the tension zone, a narrow belt that separates the more heavily forested hardwood, mixed conifer hardwood, and pine forests of northern Wisconsin from the prairies, oak savannas, and mixed deciduous forests of the south (Curtis 1959). In the south, fragmentation due to agriculture and urbanization is pronounced: almost all areas that were historically prairie and savanna have been converted to agriculture, and nearly all remaining oak-dominated woodlands and savannas have succeeded to closed canopy forests (Rhemtulla et al. 2009). The pre-settlement fire regimes for ecosystems in Wisconsin, which are believed to have been strongly influenced by Native Americans,

Table 1. Data sources used to map vegetation and calculate indices of benefit and feasibility in the analysis.

Data	Metric	Description of use	Documentation
LANDFIRE 2010	existing vegetation	discrete classification of community types	LANDFIRE; www.landfire.gov
Wisconsin Wildlife Action Plan	benefit to managing with prescribed fire	discrete classification of pixels in/out of Conservation Opportunity Areas (COPs)	Wisconsin Department of Natural Resources; http://dnr.wi.gov/topic/wildlifehabitat/actionplan.html
Wildland–urban interface (WUI) 2010	feasibility of using prescribed fire	discrete classification of pixels in/out of WUI	SILVIS Lab; Radeloff et al. (2005); http://silvis.forest.wisc.edu/maps/wui_main

Note: Information on where to access these data included, as well as references that describe methods or datasets.

ranged from frequent low-intensity fires in the southern prairies and oak savannas to infrequent stand-replacing fires in northern jack pine (*Pinus banksiana*) forests that initiated successional sequences (Anderson 1983, 1998, Schulte and Mladenoff 2005, Sands and Abrams 2011).

Data

We used nationwide, publicly available data to enable our approach to be easily replicated in other locations across the USA. To achieve our objectives, we needed spatial information on existing vegetation, areas of high ecological value, and constraints for the use of prescribed fire. We obtained information on vegetation and mean historic fire return intervals from the Landscape Fire and Resource Management Planning dataset *(available online;* Table 1).[7] Wisconsin's Wildlife Action Plan (Wisconsin DNR 2005; Table 1) identified priority areas for biodiversity conservation using an expert-based process that considered locations of high-quality natural communities, rare or declining wildlife species, and large, minimally fragmented systems, as well as priority conservation sites in other plans (Carter et al. 2014). A national wildland–urban interface assessment (Radeloff et al. 2005; Table 1) was used to identify where use of prescribed fire might be challenging because of the presence of nearby structures (Costanza et al. 2013). Wildlife Action Plan priority areas were available as polygons in GIS format; LANDFIRE and wildland–urban interface data were available as 30-m raster files in GIS format. We considered both public and private lands to provide a comprehensive view of fire management needs for a region in which fire management is conducted by federal and state agencies, non-governmental organizations (NGOs), private contractors, and private landowners. Our spatial analysis was based on the U.S. Geological Survey hydrologic unit subwatersheds (i.e., HUC12s), of which there are 1805 in Wisconsin (USGS 2009). Watersheds are a commonly used analysis unit in spatial conservation prioritizations (Margules and Pressey 2000). HUC12 units in Wisconsin range in size from 1 to 615 km^2 with an average of 80 km^2. For this study, we refer to these HUC12 subwatersheds as "management units".

Data analysis

To identify priority areas for the application of prescribed fire in Wisconsin, we followed seven steps: (1) define the prioritization goal, (2) identify frequently burned vegetation, (3) assign rarity values to vegetation, (4) estimate the benefits to be gained by applying fire, (5) estimate the management cost (i.e., effort) of applying prescribed fire, (6) estimate the feasibility of long-term fire management, and (7) combine this information to prioritize management units for application of prescribed fire. Our approach was modified from Joseph et al. (2009) to reflect local data availability, stakeholder needs, and key considerations in applying prescribed fire as a management tool.

To improve both the quality of the product and its utility for managers and policymakers, we sought input from land management agencies and conservation organizations that currently manage lands in Wisconsin using prescribed fire. Stakeholders included staff of the Wisconsin Department of Natural Resources, The Nature Conservancy, the U.S. Forest Service, and the U.S. Fish and Wildlife Service. Including these stakeholders from the beginning created a collaborative learning environment (see Roux et al. 2006) and changed our approach to meet the end user needs. We met with the stakeholder group three times to share information on anticipated data layers, methods to be used in the analysis, and preliminary results. At each meeting we answered questions and requested input on key goals, assumptions, and analysis steps. A number of decisions resulted from this input, including the use of terminology that would facilitate more effective sharing of the results with policy makers, adjustments to the LANDFIRE data, which are described in step 2, and the presentation of multiple prioritization scenarios. We also consulted with these stakeholders and with other technical experts as needed throughout the project to address specific data questions and information needs.

Step 1. Define the prioritization goal.—The specific goal of the prioritization was to maintain representation of the full suite of frequently burned, fire-dependent natural communities in Wisconsin, while minimizing management effort and maximizing management feasibility (i.e., likelihood of success).

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[7] http://landfire.cr.usgs.gov/viewer/

S*tep 2. Identify frequently burned vegetation.*—We focused on frequently burned, fire-dependent ecosystems (hereafter frequently burned ecosystems) for this analysis. We defined frequently burned ecosystems as those having historical fire return intervals of 50 yr or less. This decision was based on input from our stakeholder group: land managers were most concerned with natural communities requiring frequent fire and indicated that a 50 yr time frame was most relevant for organizational planning and budgeting activities related to land management.

Our stakeholders provided a great deal of input in the development of the community groups we used in our analysis. The existing vegetation type (hereafter vegetation types) data in LANDFIRE provided locations of vegetation, and a crosswalk of the vegetation types to the LANDFIRE biophysical settings (see footnote 1) gave us information on the mean fire return intervals (see Appendix S1: Table S1). Of the initial list of vegetation types in Wisconsin, we did not consider vegetation types for which prescribed fire is not relevant. Specifically, we excluded (1) all areas classified as developed or agriculture (e.g., orchards, row crops, fallow croplands), (2) vegetation types with a mean fire-return interval >50 yr, and (3) vegetation types with no defined fire-return interval. In addition to the decision to focus on frequently burned vegetation, stakeholders adjusted the mean fire-return intervals of six vegetation types identified in the LANDFIRE data (see Appendix S1: Table S1) to reflect local conditions and knowledge. Stakeholders requested that we include patches identified as recently burned or logged (<2% of total area of frequently burned ecosystems) based on conditions prior to disturbance (i.e., fill with surrounding vegetation type). With stakeholder input, we combined the remaining vegetation types into 11 community groups (see Appendix S1: Table S2). Groups were based on vegetation descriptions and mean fire-return intervals to represent broad ecological communities and to minimize the impacts of any misclassifications in the LANDFIRE data products. Lastly, the stakeholders requested that managed grasslands be included as part of our analysis.

Pastures and other managed grasslands were of interest to our stakeholders because they may include native prairie plants and provide habitat for grassland birds (Brennan and Kuvlesky 2005). We identified pasture by excluding pixels of the pasture/hay vegetation type that were (1) outside the historic range of prairies and savannas in Wisconsin or (2) classified as row crops in National Landcover Database (NLCD) in either 2001 or 2006. We recognize that this procedure probably overestimates the amount of land for which fire management is likely, but opted to be more inclusive so as not to overlook sites with potential conservation value.

We defined a patch of frequently burned vegetation as adjacent pixels of any type of frequently burned vegetation and removed very small patches. Although LANDFIRE products are delivered as 30-m pixels, the data is not suggested for use at the scale of individual pixels, which is why we excluded patches with fewer than four adjacent pixels (i.e., 0.0036 km^2 or 0.9 acres). We selected this cut-off point based on the input of stakeholders, who considered sites less than one acre in size to be too small to be suitable as a focus of long-term management efforts in a statewide analysis.

Step 3. Assign rarity rankings.—We assigned the Wisconsin Natural Heritage Inventory rarity rankings (Wisconsin DNR 2014) to the 12 community groups. For each of the groups, we identified the highest rank (state or global) of the vegetation types included (see Appendix S1: Table S2). We assigned a rarity value to the 12 community types by reversing the Natural Heritage Inventory scale (i.e., 6-rarity rank value), which gave the most imperiled communities the numerically largest values.

Step 4. Estimate benefits of fire management.—We calculated the management benefit of each management unit. First, we calculated the proportion of the total area of each community group present in the state that occurred within the boundary of each management unit to estimate the importance of the individual management unit for maintaining representation of each community type in the state. Next, we weighted more heavily the areas of frequently burned vegetation that overlapped with identified statewide conservation priority areas (see Appendix S2: Fig. S1). The weight applied to each community group in each management unit reflected the proportion of the area of the community group overlapping with Wisconsin Wildlife Action Plan conservation priority areas in that management unit. The overall management benefit index for the management unit was then the sum of the expected benefit of managing each community type in the management unit (Eq. 1). Values of management benefit ranged from 0 to 1.8. There was only one management unit with a benefit index value greater than 1.0, so we did not scale these values.

$$\begin{aligned}&\text{Management benefit index}_{\text{Management Unit (MU)}}\\&\quad=\sum_{i=1}^{n}[\text{Proportion of statewide area of}\\&\quad\text{community group}_i\text{ in MU}\times(1+\text{Proportion of}\\&\quad\text{the area of community group}_i\text{ in MU that}\\&\quad\text{overlaps a priority area})]\end{aligned} \tag{1}$$

Step 5. Estimate management effort.—The application of prescribed fire is staff-, time-, and equipment-intensive. For a number of reasons, stakeholders did not want us to estimate dollar values for managing frequently burned vegetation in the state using prescribed fire. Together with stakeholders, we identified an alternate index, called management effort, that would be more helpful for stakeholders and still accurately reflect the resources needed to manage frequently burned vegetation. We calculated the management effort index for each management unit by calculating the number of prescribed fires that would need to be conducted (based on the mean fire-return interval of the dominant community type in each patch) to manage all patches of frequently burned vegetation within the management unit over a 50 yr time period (Eq. 2). Thus, the management effort index is essentially a function of the level of fragmentation of fire-dependent vegetation in the management unit. To scale these values to a range that was similar to those for the benefit index, we divided all management effort index values by the maximum index value for a management unit, resulting in management index values that ranged from zero to one.

$$\text{Management effort index}_{\text{Management Unit}} = \sum_{k=1}^{n} \text{Number of patches of fire dependent vegetation with dominant fire return interval}_k \times \left(\frac{50}{\text{fire return interval}_k}\right) \quad (2)$$

Step 6. Estimate management feasibility.—We defined the management feasibility index as the likelihood that managers would be able to successfully apply prescribed fire to a patch of frequently burned vegetation. The feasibility index was a combination of the probability that prescribed fires would successfully maintain the ecological values of the communities contained in the patch and the probability that prescribed fire could be actually be applied successfully to the patch (Eq. 3). For simplicity, we assumed that the former is one (i.e., if the patch is burned at the historical mean fire return interval, the ecological values of the communities within the patch will persist). To estimate the probability of successfully applying prescribed fire to the patch, we considered whether the patch overlapped with the wildland–urban interface (WUI; see Appendix S2: Fig. S2). We set the probability that it is possible to successfully apply prescribed fire to a patch that had any overlap with the WUI to 50% based on discussions with stakeholders. Their consensus was that in such patches, conditions that are favorable for conducting prescribed fires occur only half as often (e.g., due to weather limitation) than in patches that do not overlap with the WUI. In addition, there are also important reasons to target WUI areas for management (e.g., reducing the risk of fire to human structure and communities). Stakeholders felt that a value of 50% adequately balanced these considerations in Wisconsin. This index is essentially a function of how many patches overlap with WUI, and the feasibility index value for a management unit decreases as WUI overlap increases. Similar to the values for management effort, we scaled the management feasibility index values from zero to one.

$$\text{Management feasibility index}_{\text{Management Unit}} = 1 - (\text{proportion of patches of fire dependent vegetation that overlap with the WUI} \times 0.5) \quad (3)$$

Step 7. Identify management priorities.—We used community rarity values and the indices of management benefit, management effort, and management feasibility to prioritize the application of prescribed fire under three different scenarios: maximum ecological benefit, maximum ecological benefit with minimum effort, and a comprehensive prioritization (maximum ecological benefit with minimum effort and maximum feasibility; Fig. 1). This flexible approach to prioritizing prescribed fire effort and these specific endpoints reflect input from the stakeholder group. Stakeholders wanted to compare the calculated management benefit and ecological benefit, allowing them to visually assess the influence of including a community rarity ranking. Presenting both scenarios also allows stakeholder to choose the approach that most closely fits with their agency's or organization's mission. Similarly, stakeholders wanted prioritization scenarios with or without management feasibility and effort indices, so that individual user groups could choose the scenario most relevant to their organization's mission, funding sources, land ownership and management responsibilities, and prescribed fire capacity. Stakeholders felt like this approach of presenting multiple prioritization scenarios was more informative for them (and the policy makers with whom they work) than a formal sensitivity analysis.

The ecological benefits scenario (Eq. 4) prioritized communities identified by the Natural Heritage Inventory as imperiled or critically imperiled, and deemphasized managed grasslands (e.g., pastures). The maximum ecological benefit with minimum effort scenario (Eq. 5) prioritized areas where the application of prescribed fire would have the greatest benefits for rare community types dependent on fire with the least effort. Lastly, the comprehensive prioritization (Eq. 6) identified where prescribed fire has the greatest ecological benefits, requires the least effort, and offers the greatest feasibility for long-term land management.

$$\text{Ecological benefit prioritization} = \text{Community Rarity} \times \text{Management Benefit index} \quad (4)$$

$$\text{Maximum ecological benefit with minimum effort prioritization} = \frac{\text{Community Rarity} \times \text{Management Benefit index}}{\text{Effort index}} \quad (5)$$

$$\text{Comprehensive prioritization} = \frac{\text{Community Rarity} \times \text{Management Benefit index} \times \text{Feasibility Index}}{\text{Effort index}} \quad (6)$$

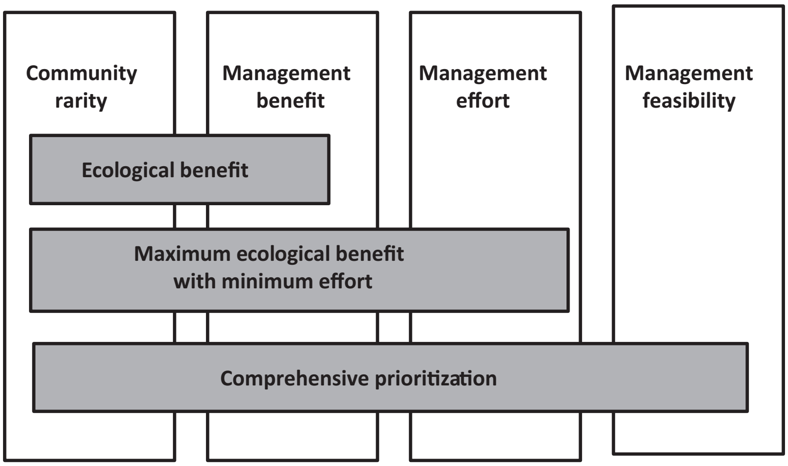

FIG. 1. Conceptual diagram of how rarity and indices are combined to create the prioritizations. Rarity and indices (benefit, effort, and feasibility) are shown as columns. Prioritizations (ecological benefit, maximum ecological benefit with minimum effort, and comprehensive) are shown as rows that overlap with the indices combined to calculate the prioritization.

RESULTS

We estimated that the area that would benefit from fire management as defined for our analysis is 18153 km^2, which is approximately 13% of the state. Totaling the annual fire needs of each community group (i.e., area retained divided by fire return interval; Table 2), suggests that roughly 4300 km^2 would need to be burned annually to maintain frequently burned vegetation in the state of Wisconsin.

Of this total area of frequently burned vegetation, we found considerable variation in the area of individual community groups. Very little dry prairie and tallgrass prairie remain in Wisconsin (less than 150 km^2 each), and remnant prairies often occur in very small patches (Wolf 2004, Hotchkiss et al. 2007). The areas of oak savanna, dry prairie, oak barrens, and bluff and talus habitats were also under 1000 km^2 (Table 2). Woody-dominated systems with longer fire-return intervals (>10 yr), which included the pine-oak forest and pine forest, represented 12.5% of the frequently burned vegetation in the state, while those with shorter fire-return intervals (<10 yr), oak woodland, oak forest, and pine barrens, made up almost 45% of the total frequently burned vegetation. Herbaceous wetlands were second in area to managed grasslands among the

TABLE 2. We identified 12 community groups for our analysis, each having been assigned a historical mean fire-return interval. Here we show the total area and number of patches before and after filtering out small patches (fewer than four adjacent pixels) and the percent area or patches retained for analysis.

Community groups	Historical mean fire–return interval	Total area (km^2)	Area retained (km^2)	Area retained (%)	Total patches	Number of patches retained	Patches retained (%)
Tallgrass prairie	3	127	107	85%	39,682	21,656	55%
Oak savanna	4	132	120	91%	31,151	20,076	64%
Dry prairie	3	170	144	85%	47,498	23,228	49%
Oak barrens	5	484	417	86%	1,14,844	51,248	45%
Bluff and talus	4	1,202	1,078	90%	2,17,775	1,02,364	47%
Pine-oak forest	26	1,413	1,083	77%	4,28,142	1,26,372	30%
Pine forest	12	1,502	1,180	79%	4,02,112	1,17,270	29%
Herbaceous wetlands	3	2,283	1,910	84%	4,38,687	1,29,311	29%
Oak woodland	5	2,347	2,121	90%	3,77,179	1,68,343	45%
Oak forest	5	3,226	2,870	89%	5,12,085	1,96,586	38%
Pine barrens	4	3,427	3,169	92%	3,46,175	1,09,931	32%
Managed grasslands	3	4,362	3,954	91%	5,42,138	2,12,029	39%

Note: Community groups are ranked from least to greatest area retained (km^2).

herbaceous-dominated systems (Table 2). Overall, we retained at least 77% of the area of each community type for analysis, although the percentage of patches retained ranged from 29% to 64% (Table 2).

Differences in vegetation across the state resulted in strong spatial patterns of frequently burned communities. The majority of the frequently burned vegetation was south of the tension zone, an area which historically had shorter fire return intervals (Fig. 2A) due to climate, soils, and historically larger populations of Native Americans (Curtis 1959, Dorney 1981). We identified less frequently burned vegetation north of the tension zone because most of this area consists of forests with historical fire-return intervals greater than 50 yr. However, we did capture the frequently burned pine and oak barrens (Curtis 1959) that occupy droughty soils in parts of northwestern and northeastern Wisconsin (Fig. 2A). Community groups that we identified as imperiled and critically imperiled, including prairie and savanna community groups, occurred both north and south of the tension zone, with areas of high concentration in the northwest and central regions where remnant prairies and oak barrens are located (Fig. 2B). Whereas the majority of the secure managed grasslands occurred in the southwestern portion of the state (Fig. 2B).

The individual indices demonstrated that the benefits to managing the landscape with prescribed fire were widespread across Wisconsin, but that management effort increases when frequently burned communities are fragmented. Management units with high ecological benefits occurred predominantly south of the tension zone and in two distinct regions in the northwest and northeast portion of the state (Fig. 3A), reflecting the location of rare community types and existing biodiversity conservation priority areas. Management effort was higher in management units south of the tension zone (Fig. 3B) where patches of frequently burned vegetation are typically smaller and dominated by community types with shorter fire-return intervals. Management feasibility was higher in units south of the tension zone, especially southwest Wisconsin (Fig. 3C). Management feasibility was lowest in the forested northern portion of the state and in the southeast (Fig. 3C), attributable to a high proportion of WUI occurring in these areas (see Appendix S2: Fig. S2).

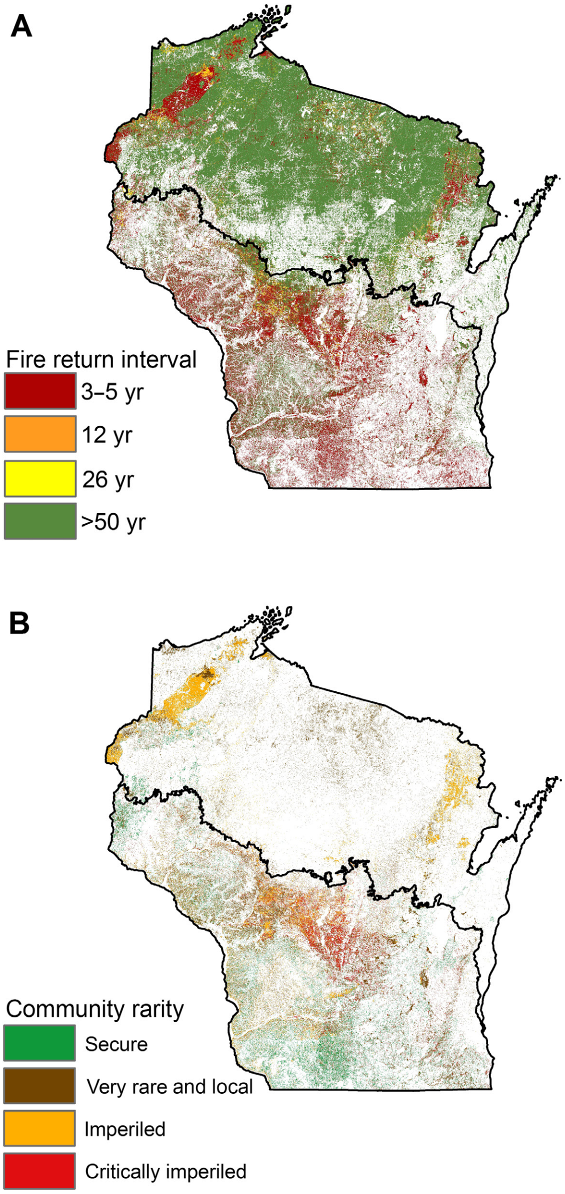

FIG. 2. Maps of fire return interval and rarity ranking for community groups in the State of Wisconsin, USA. Vegetation grouped by the (A) historic mean fire-return interval and ranges from most frequently burned (3–5 yr) to fire-return intervals greater than 50 yr. For our analysis, we only used frequently burned vegetation with a mean fire-return interval of less than 50 yr, thereby excluding the majority of the vegetation in the northern portion of Wisconsin. For community groups included in the analysis, we assigned (B) rarity rankings based on Natural Heritage Inventory (NHI) documentation. Although the NHI scale has five categories, for simplicity we combine the two secure categories into a single color (green) on this map. Each map is overlaid with a line representing the tension zone.

Management units that ranked as priority areas for long-term management with prescribed fire occurred throughout Wisconsin. Management units with high ecological benefit were more abundant south of the tension zone (Fig. 4A), attributable to both existing biodiversity conservation priority areas and rarity of the community types in the southern portion of the state. The ranking of management units of maximum ecological benefit with minimum effort was similar to the ecological rankings, with high priority areas in the central region, northwest, and northeast (Fig. 4B). This similarity is likely the result of similar management effort values in high ecological benefit areas. The majority of the management units identified as the greatest priority based on our comprehensive ranking occurred south of the tension zone (Fig. 4C), where prairies and savannas historically occurred. The imperiled barren communities of the northwest and northeast were also identified as high priority management units (Fig. 4C).

DISCUSSION

We identified a substantial need for prescribed fire as a land management tool across the State of Wisconsin. Comparing estimates of prescribed fire applied in Wisconsin from 2004 to 2009, which range from 140 to 200 km^2/year (T. Trapp, *personal communication*), to the estimate from our results (4300 km^2/year) demonstrates that the area requiring action is far greater than the resources available. Given that many approaches to identifying areas for conservation and protection do not take into account the need for long term land management of sites, we feel that this analysis sets clear priorities for land management action where resources are limited. In using a cost–benefit approach, we sought to balance expected benefits with expected costs while considering the practical feasibility of managing areas using prescribed fire.

The spatially explicit output of our analysis demonstrates the widespread geographic need for managing frequently burned ecosystems in Wisconsin. The majority of the community groups identified for management with prescribed fire were the rare prairie, savanna, and barren ecosystems. Given the potential for rapid encroachment of grasslands and savannas by woody plants (Lehman et al. 2014), these communities represent an important priority for conservation action in the upper Midwest (Briggs et al. 2005, Wisconsin DNR 2014) and worldwide (Scholes and Archer 1997, Myers et al. 2000). The wide geographic distribution of ecosystems that require frequent fire across the state reinforces the need for statewide planning and for identifying mechanisms to assist landowners in managing rare, fire-dependent community types occurring on private lands.

The indices that we developed to reflect management benefits, effort, and feasibility were simple and informative. We used the Wildlife Action Plan as part of our benefit index because existing conservation priority areas matter when land managers decide upon management actions (Game et al. 2013), and this decision was encouraged by our stakeholders. Similarly, our use of the existing WUI data incorporated the inherent challenges in using prescribed fires to manage fire-dependent natural communities in areas that are susceptible to wildfire and pose risk to homes (Dellasala et al. 2004, Radeloff et al. 2005). Taken alone, the indices of benefit, effort, and feasibility may be useful for more fine-scale analyses to determine barriers to prescribed fire use.

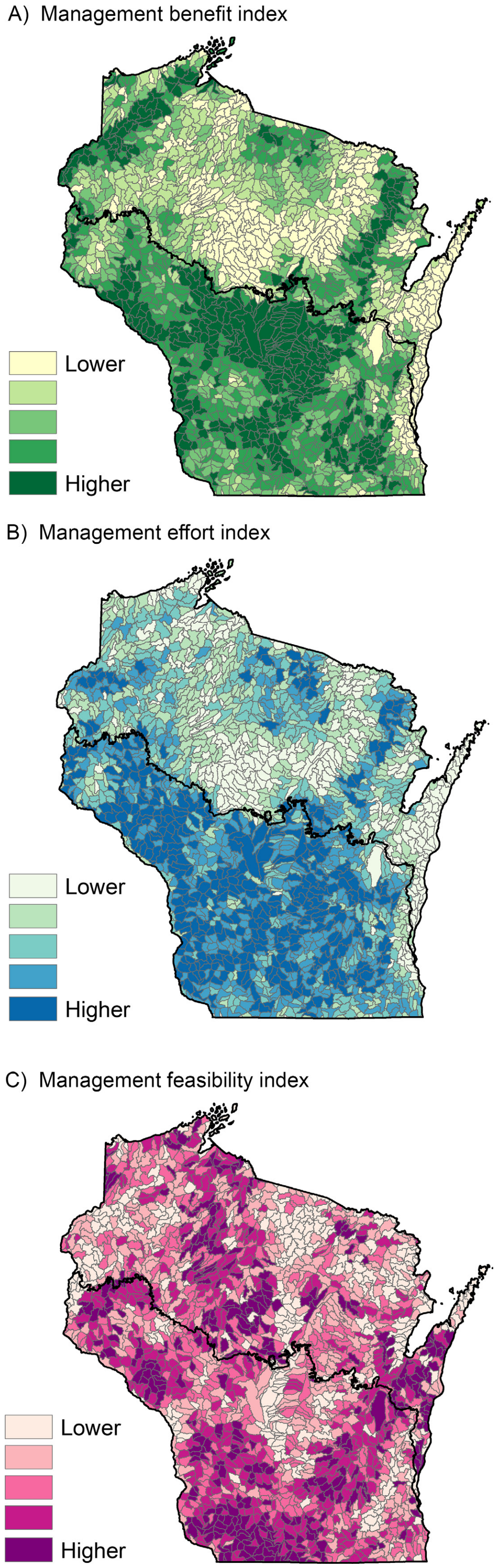

FIG. 3. Maps of three indices of (A) management benefit, (B) management effort, and (C) management feasibility, are summarized as 20% quantiles by management units (HUC12 subwatersheds). Each map is overlaid with a line representing the tension zone.

Our approach of showing multiple indices and prioritization schemes provided our stakeholders, and other end-users, maps of priority areas that can be used for different planning purposes at a landscape scale. Often, prescribed fire planning takes place on a relatively small scale identifying the priorities on individual properties (e.g., Hiers et al. 2003), but having a statewide prioritization map can help to put the needs of individual properties into a larger context. For example, the statewide analysis may determine which large properties (e.g., state forest) that are within high benefit regions (Hiers et al. 2003, Costanza and Moody 2011), or to identify barriers to the implementation of prescribed fire (Costanza et al. 2013) in high management benefit areas that are currently not being actively managed. Similarly, landscapes that could benefit from prescribed fire but fall within low feasibility management units may reveal opportunities for targeted public education and outreach about the benefits of prescribed fire.

The ability to extract each of the individual indices that were combined to produce the final prioritization allowed users to understand how the various components contributed to the final product and helped to build stakeholder buy in. All three prioritization schemes show high priority areas for prescribed fire management in several distinct regions, which suggests that trained staff and wildland fire equipment need to be strategically located there (Dumoulin et al. 2014). The comprehensive ranking map may also be used identify regions where the various agencies managing frequently burned communities can more effectively combine resources to apply prescribed fire (Goldstein and Butler 2010). Maximizing ecological benefits with minimum management effort may be useful when there are severe limitations placed upon the budgets and staff of conservation organizations.

Our approach of prioritizing sites for prescribed fires on both public and private lands resulted in a more comprehensive analysis of the state and is important for identifying land management opportunities. Currently in the USA, the focus of wildland fire policy is on reducing hazards and maintaining ecosystems (Dellasala et al. 2004). However, when prescribed burns only occur on public lands, there are missed opportunities for conservation and hazard reduction on private lands. Although private landowners can be hesitant to use prescribed fire (Morton et al. 2010), the success of private landowner burning cooperatives and prescribed burn associations in Nebraska (Twidwell et al. 2013) and Texas (Toledo et al. 2014) could serve as a model for increasing the use of prescribed fire on fire-dependent natural

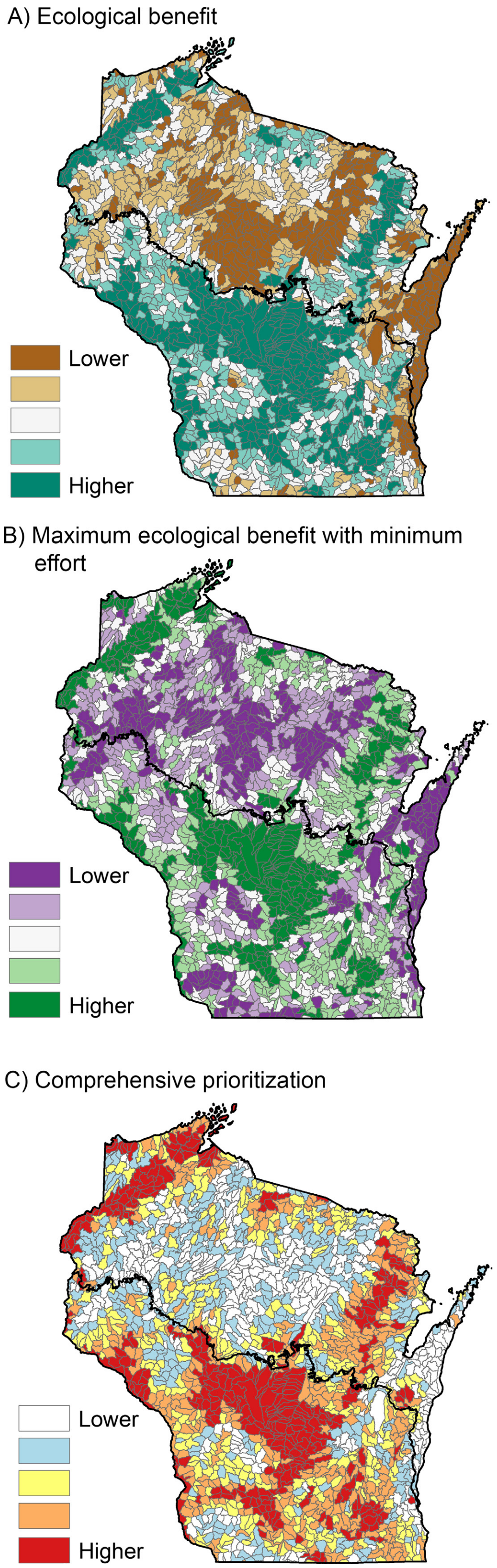

FIG. 4. Maps of statewide prioritizations for management with prescribed fire. Prioritizations include the (A) ecological benefit, (B) maximum ecological benefit with minimum effort, and (C) comprehensive prioritization. Data are summarized as 20% quantiles by management units (HUC12 subwatersheds), and each map includes a line to represent the tension zone.

communities located on private land. Similarly, effective management of fire-dependent communities within the wildland–urban interface (WUI) may require education or subsides to encourage private landowners to apply fuel treatments to more effectively mitigate the negative impacts of future wildfires (Schoennagel et al. 2009, Moritz et al. 2014).

While our study was overall successful in meeting stakeholder needs, we recognize several limitations to the analysis. First, we focused on management of fire-dependent communities using only prescribed fire. Prescribed fire is an important tool for managing communities that depend on frequent fire and is used by both agencies and conservation organizations across the USA (Dellasala et al. 2004) and elsewhere, but it is not the only way in which the ecological value of grassland and savanna communities can be maintained. In the Midwest, mowing and grazing are also sometimes used to manage communities historically structured by fire (MacDougal and Turkington 2007, Begay et al. 2011). Our analysis could also be used to identify locations where management of fire-dependent communities is a high priority, but where other tools (e.g., grazing) may be more desirable because the feasibility of applying prescribed fire is low.

Our focus on prescribed fire for conservation is also a simplification of wildland fire management, and our model does not include fire severity, wildfires, or strategies to protect resources. We assume that prescribed fires will meet burn objectives, but realize this is not always the case and that prescribed fires can have unintended ecological consequences (e.g., high-severity fires killing more overstory trees than intended). In regions where natural ignitions can be used to meet management objectives, we suggest that using them might be preferable, and the identified high priority regions for prescribed fire management could be used as the starting point for identifying where managed wildfires could be used. Our method could also be modified for other more specific uses, for example, to give greater priority to management action within the WUI as a means of reducing fire risk to human structures, while simultaneously achieving desired ecological benefits.

Second, the indices that we developed for the benefit, effort, and feasibility of applying prescribed fire were specific to the available data and issues associated with using prescribed fire as a management tool. In principle, our methods can easily accommodate additional or alternate data on the benefit, effort, cost, or feasibility of management tailored to the region or management action of interest. Thus, we suggest that our work is relevant to the broader land management community as natural resource managers seek to prioritize their land management efforts where management is most feasible and will have the greatest conservation benefits (Naidoo et al. 2006, Sundell-Turner and Rodewald 2008, Ager et al. 2013).

Conclusions

We identified priority areas for managing frequently burned vegetation in Wisconsin using a cost–benefit approach that considered benefits, effort, and feasibility of management with prescribed fire. Our approach allows end users to consider ecological goals, such as working within existing conservation priority areas, along with practical limitations given management capacities and the likelihood of being able to successfully manage areas in the long term. Our approach provided a great level of transparency and produced a product that can be easily updated as new data become available. While we focused on prescribed fire as a management action, the methods presented here can be used to identify priority areas for other land management activities, including fuel reduction treatments or herbicide to treat invasive species, as long as planners are able to identify benefits, costs, and feasibility associated with a given action.

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Literature Cited

Ager, A. A., N. M. Vaillant, and A. McMahan. 2013. Restoration of fire in managed forests: a model to prioritize landscapes and analyze tradeoffs. Ecosphere 4:19.

Anderson, R. C. 1983. The eastern prairie-forest transition: an overview. Pages 86–92 *in* R. Brewer, editor. Proceedings of the 8th North American Prairie Conference. Western Michigan University, Kalamazoo, Michigan, USA.

Anderson, R. C. 1998. Overview of Midwestern oak savanna. Transactions of the Wisconsin Academy of Sciences, Arts, and Letters 86:1–18.

Begay, B., H. M. Alexander, and E. Questad. 2011. Effects of mid-summer haying on growth and reproduction in prairie forbs. Transactions of the Kansas Academy of Science 114:108–114.

Brennan, L. A., and W. P. Kuvlesky. Morrison. 2005. Invited paper: North American grassland birds: an unfolding conservation crisis? Journal of Wildlife Management 69:1–13.

Briggs, J. M., A. K. Knapp, J. M. Blair, J. L. Heisler, G. A. Hoch, M. S. Lett, and J. K. McCarron. 2005. An ecosystem in transition: causes and consequences of the conversion of mesic grassland to shrubland. BioScience 55:243–254.

Brooks, T. M., R. A. Mittermeier, G. A. B. da Fonseca, J. Gerlach, M. Hoffman, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. Science 313:58–61.

Carter, S. K., J. D. Pohlman, T. L. Bergeson, C. M. Hamilton, A. M. Pidgeon, and V. C. Radeloff. 2014. Adding value to existing conservation plans using projected housing growth. Landscape and Urban Planning 126:10–20.

Carwardine, J., T. O'Connor, S. Legge, B. Mackey, H. P. Possingham, and T. G. Martin. 2012. Prioritizing threat management for biodiversity conservation. Conservation Letters 5:196–204.

Collins, B. M., S. L. Stephens, J. J. Moghaddas, and J. Battles. 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry 108:24–34.

Commission for Environmental Cooperation. 1997. Ecological regions of North America: toward a common perspective. Commission for Environmental Cooperation, Montreal, Quebec, Canada. Map (scale 1:12,500,000). Revised 2006. http://www.epa.gov/wed/pages/ecoregions.htm.

Costanza, J. K., and A. Moody. 2011. Deciding where to burn: stakeholder priorities for prescribed burning of a fire-dependent ecosystem. Ecology and Society 16:14.

Costanza, J. K., J. Weiss, and A. Moody. 2013. Examining the knowing-doing gap in the conservation of a fire-dependent ecosystem. Biological Conservation 158:107–115.

Curtis, J. T. 1959. Vegetation of Wisconsin. University of Wisconsin Press, Madison, Wisconsin, USA.

Dellasala, D. A., J. E. Williams, C. D. Williams, and J. F. Franklin. 2004. Beyond smoke and mirrors: a synthesis of fire policy and science. Conservation Biology 18:976–986.

Dorney, J. R. 1981. The impact of native Americans on presettlement vegetatin in southeastern Wisconsin. Wisconsin Academy of Sciences, Arts and Letters 69:26–36.

Dumoulin, C. E., T. Macmillan, R. Stoneman, and P. R. Armsworth. 2014. Locating human resources to reduce the cost of managing networks of protected areas. Conservation Letters 7:553–560.

Game, E. T., P. Kareiva, and H. P. Possingham. 2013. Six common mistakes in conservation priority setting. Conservation Biology 27:480–485.

Goldstein, B. E., and W. H. Butler. 2010. Expanding the scope and impact of collaborative planning. Journal of the American Planning Association 76:238–249.

Hessburg, P. F., K. M. Reynolds, R. E. Keane, K. M. James, and R. B. Salter. 2007. Evaluating wildland fire danger and prioritizing vegetation and fuels treatments. Forest Ecology and Management 247:1–17.

Hesseln, H. 2000. The economics of prescribed burning: a research review. Forest Science 46:322–334.

Hiers, J. K., S. C. Laine, J. J. Bachant, J. H. Furman, J. W. W. Greene, and V. Compton. 2003. Simple spatial modeling tool for prioritizing prescribed burning activities at the landscape scale. Conservation Biology 17:1571–1578.

Hoffmann, W. A., and O. T. Solbrig. 2003. The role of topkill in the differential response of savanna woody species to fire. Forest Ecology and Management 180:273–286.

Hotchkiss, S. C., R. Calcote, and E. A. Lynch. 2007. Response of vegetation and fire to Little Ice Age climate change: regional continuity and landscape heterogeneity. Landscape Ecology 22:25–41.

Joseph, L. N., R. F. Maloney, and H. P. Possingham. 2009. Optimal allocation of resources among threatened species: a project prioritization protocol. Conservation Biology 23:328–338.

Keane, R. E., S. A. Drury, E. C. Karau, P. F. Hessburg, and K. M. Reynolds. 2010. A method for mapping fire hazard and risk across multiple scales and its application in fire management. Ecological Modelling 221:2–18.

Knight, A. T., R. M. Cowling, M. Difford, and B. M. Campbell. 2010. Mapping human and social dimensions of conservation opportunity for scheduling of conservation action on private land. Conservation Biology 24:1348–1358.

Lehman, C. E. R., et al. 2014. Savanna vegetation-fire-climate relationships differ among continents. Science 343:548–552.

MacDougal, A. S., and R. Turkington. 2007. Does the type of disturbance matter when restoring disturbance-dependent grasslands? Restoration Ecology 15:263–272.

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

Moilanen, A., K. A. Wilson and H. P. Possingham. 2009. Spatial conservation prioritization: quantitative methods and computational tools. Oxford University Press, Oxford, UK.

Moritz, M. A., et al. 2014. Learning to coexist with wildfire. Nature 515:58–66.

Morton, L. W., E. Regen, D. M. Engle, J. R. Miller, and R. N. Harr. 2010. Perceptions of landowners concerning conservation, grazing, fire, and eastern red cedar management in tallgrass prairie. Rangeland Ecology and Management 63:645–654.

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

Naidoo, R., A. Balmford, P. J. Ferraro, S. Polasky, T. H. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. Trends in Ecology and Evolution 21:681–687.

Peet, R. K. and D. J. Allard. 1993. Longleaf pine vegetation of the Southern Atlantic and Eastern Gulf Coast regions: a preliminary classification. Pages 45-81 *in* S. M. Herman, editor. Proceedings of the Tall Timbers Fire Ecology Conference, Number 18. The longleaf pine ecosystem: ecology, restoration and management.

Pendergrass, K. L., P. M. Miller, J. B. Kauffman, and T. N. Kaye. 1999. The role of prescribed burning in mantenane of an endangered plant species, *Lomatium bradshawii*. Ecological Applications 9:1420–1429.

Peterson, D. W., and P. B. Reich. 2001. Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. Ecological Applications 11:914–927.

Platt, W. J. 1999. Southeastern pine savannas. Pages 23–51 *in* R. C. Anderson, J. S. Fralish, and J. M. Baskin, editors. Savannas, barrens, and rock outcrop plant communities of North America. Cambridge University Press, Cambridge, UK.

Quinn-Davidson, L. N., and J. M. Varner. 2011. Impediments to prescribed fire across agency, landscape and manager: an example from northern California. International Journal of Wildland Fire 21:210–218.

Radeloff, V. C., R. B. Hammer, S. I. Stewart, J. S. Fried, S. S. Holcomb, and J. F. McKeefry. 2005. The wildland–urban interface in the United States. Ecological Applications 15:799–805.

Rhemtulla, J. M., D. J. Mladenoff, and M. K. Clayton. 2009. Legacies of historical land use on regional forest composition and structure in Wisconsin, USA (mid-1800s–1930s–2000s). Ecological Applications 19:1061–1078.

Roux, D. J., K. H. Rogers, H. C. Biggs, P. J. Ashton, and A. Sergeant. 2006. Bridging the science–management divide: moving from unidirectional knowledge transfer to knowledge interfacing and sharing. Ecology and Society 11:4. www.ecologyandsociety.org/vol11/iss1/art4

Salafsky, N., R. Margoluis, K. H. Redford, and J. G. Robinson. 2002. Improving the practice of conservation: a conceptual framework and research agenda for conservation science. Conservation Biology 16:1469–1479.

Sands, B. A., and M. D. Abrams. 2011. A 183-year history of fire and recent fire suppression impacts in select pine and oak forest stand of the Menominee Indian Reservation, Wisconsin. American Midland Naturalist 166:325–338.

Schoennagel, T., C. R. Nelson, D. M. Theobald, G. C. Carnwath, and T. B. Chapman. 2009. Implementation of National Fire Plan treatments near the wildland–urban interface in the western United States. Proceedings of the National Academy of Sciences 106:10706–10711.

Scholes, R. J., and S. R. Archer. 1997. Tree–grass interactions in savannas. Annual Review of Ecology and Systematics 28:517–544.

Schulte, L. A., and D. J. Mladenoff. 2005. Severe wind and fire regimes in northern forests: historical variability at the regional scale. Ecology 86:431–445.

Scott, J. M., D. D. Goble, J. A. Wiens, D. S. Wilcove, M. Bean, and T. Male. 2005. Recovery of imperiled species under the Endangered Species Act: the need for a new approach. Frontiers in Ecology and the Environment 3:383–389.

Shindler, B. A., E. Toman, and S. M. McCaffrey. 2009. Public perspectives of fire, fuels, and the Forest Service, in the Great Lakes Region: a survey of citizen-agency communication and trust. International Journal of Wildland Fire 18:157–164.

Sundell-Turner, N. M., and A. D. Rodewald. 2008. A comparison of landscape metrics for conservation planning. Landscape and Urban Planning 86:219–225.

Tear, T. H., B. N. Stratton, E. T. Game, M. A. Brown, C. D. Apse, and R. R. Shirer. 2014. A return-on-investment framework to identify conservation priorities. Biological Conservation 173:42–52.

Thompson, M. P., D. E. Calkin, M. A. Finney, A. A. Ager, and J. W. Gilbertson-Day. 2011. Integrated national-scale assessment of wildfire risk to human and ecological values. Stochastic Environmental Research and Risk Assessment 25:761–780.

Toledo, D., U. P. Kreuter, M. G. Sorice, and C. A. Jr Taylor. 2014. The role of prescribed burn associations in the application of prescribed fires in rangeland ecosystems. Journal of Environmental Management 132:323–328.

Twidwell, D., W. E. Rogers, S. D. Fuhlendorf, C. L. Wonkka, D. M. Engle, J. R. Weir, U. P. Kreuter, and C. A. T. Jr. 2013. The rising Great Plains fire campaign: citizens' response to woody plant encroachment. Frontiers in Ecology and the Environment 11:e64–e71.

U.S. Geological Survey and U.S. Department of Agriculture, Natural Resources Conservation Service. 2009. Federal guidelines, requirements, and procedures for the national Watershed Boundary Dataset. U.S. Geological Survey Techniques and Methods 11–A3, U.S. Geological Survey, Reston, Virginia, USA.

Wade, D. D., and J. D. Lundsford. 1989. A guide for prescribed fire in southern forests. USDA Forest Service Technical Publication R8-TP 11. USDA Forest Service, Atlanta, Georgia, USA.

Werner, P. A., and L. D. Prior. 2013. Demography and growth of subadult savanna trees: interactions of life history, fire season, and grassy understory. Ecological Monographs 83:67–93.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritizing global conservation efforts. Nature 440:337–340.

Wilson, K. A., et al. 2007. Conserving biodiversity efficiently: what to do, where, and when. PLoS Biology 5:e223.

Wisconsin Department of Natural Resources. 2005. Wisconsin's strategy for wildlife species of greatest conservation need. Wisconsin Department of Natural Resources, Madison, Wisconsin, USA. http://dnr.wi.gov/topic/wildlifehabitat/actionplan.html

Wisconsin Department of Natural Resources. 2014. Natural Heritage Inventory working list. Wisconsin Department of Natural Resources, Madison, Wisconsin, USA.http://dnr.wi.gov/topic/NHI/documents/NHIWorkingList.pdf

Withey, J. C., et al. 2012. Maximizing return on conservation investment in the conterminous USA. Ecology Letters 15:1249–1256.

Wolf, J. 2004. A 200-year fire history in a remnant oak savanna in southeastern Wisconsin. American Midland Naturalist 152:201–213.

Wood, D. B. 1988. Costs of prescribed burning in southwestern ponderosa pine. Western Journal of Applied Forestry 3:115–119.

Supporting Information

Additional supporting information may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1890/15-0509.1/suppinfo

Data Availability

Data associated with this paper have been deposited in Dryad: http://dx.doi.org/10.5061/dryad.s18gk