Habitat Assessment and Conservation Status of Endangered Northeastern Bulrush

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Abstract - *Scirpus ancistrochaetus* (Northeastern Bulrush) is a federally endangered sedge that grows in temporary wetlands. We performed surveys of 90 wetlands in Pennsylvania, Maryland, West Virginia, and Virginia, measuring areal extent, stem density, and number of flowering stems of Northeastern Bulrush. We also measured percentage of tree canopy closure, presence of threats, and size of wetland. Percentage of tree canopy closure was negatively correlated with wetland area, percentage of wetland area occupied by Northeastern Bulrush, total number of stems, stem density, and percentage of flowering stems. Wetland area was positively related to percentage of flowering stems and had a tendency to be positively related to stem density, likely in part due to larger wetlands having lower tree canopy closure. Invasive *Phalaris arundinacea* (Reed Canarygrass) and *Microstegium vimineum* (Japanese Stiltgrass) were present at 7% and 21% of the wetlands, respectively. *Odocoileus virginianus* (White-tailed Deer) and *Ursus americanus* (Black Bear) damage were present in 38% and 17% of wetlands, respectively. Modification of habitat was noted at 27% of wetlands. For wetlands with previous data on population size, 14% had increased, 34% were stable, 25% had decreased, and 27% were absent or had severely decreased. Our recommendations for management include reducing tree canopy closure with control of invasive species and White-tailed Deer where needed.

Introduction

The US Endangered Species Act uses sound scientific principles to enhance recovery of species threatened by extinction (NRC 1995). Basic research on a species' population biology is necessary to integrate into recovery plans for effective conservation (Schemske et al. 1994). Conservation of plants generally receives less attention than animals. Compared to those for animals, recovery plans for plants are more likely to fail to address research on species biology and to properly consider threat mitigation (Schultz and Gerber 2002). Those species with readily mitigable threats, such as those that can be addressed with ecological management, may be those that are most likely to recover (Abbitt and Scott 2001). In a review of recovery plan implementation, only about half included monitoring the results of management activities (Boersma et al. 2001). Yet continued monitoring of populations and threats is necessary for effective conservation and adaptive management in plants (MacKenzie and Keith 2009). Those recovery plans that incorporate "explicit and dynamic science" are more likely to be successful (Boersma et al. 2001), pointing towards the need for continual basic research of endangered species and re-visitation of recovery plan goals.

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Scirpus ancistrochaetus Schuyler (Northeastern Bulrush) is a perennial emergent sedge, generally found in small depressional wetlands within forested ecosystems. While some authors do not recognize Northeastern Bulrush as a species (e.g., Gleason and Cronquist 1991), others do (e.g., Kartesz and Kartesz 1980, ITIS 2010). Schuyler (1962) provided the first description of the species. Northeastern Bulrush is limited to ≈120 populations in the northeastern United States (USFWS 2009; R. Popp, VT Department of Fish and Wildlife, pers. comm.) and is currently listed as federally endangered (USFWS 1991). Northeastern Bulrush can be found in a single isolated wetland or found in one to several wetlands within a clustered wetland complex. The species is found in Maryland, Massachusetts, New Hampshire, Vermont, Virginia, and West Virginia, but most populations (57%) occur in Pennsylvania. Common habitat associates include *Glyceria canadensis* (Mich.) Trin. (Rattlesnake Mannagrass), *Cephalanthus occidentalis* L. (Buttonbush), *Ilex verticillata* (L.) A. Gray (Common Winterberry), *Dulichium arundinaceum* (L.) Britton (Threeway Sedge), and *Glyceria acutiflora* Torr. (Creeping Mannagrass) (USFWS 1993). While some wetland habitats that support Northeastern Bulrush in the northern range of this species (i.e., New Hampshire, Massachusetts, and Vermont) are similar to those in the southern range (i.e., Maryland, Pennsylvania, Virginia, and West Virginia), there are enough differences to warrant separating the two groups of wetlands into different studies. For example, wetlands in the northern range tend to be larger, are often influenced by *Castor canadensis* Kuhl (Beaver) activities, and have Northeastern Bulrush populations that fluctuate more dramatically in size (USFWS 2009; K.A Cipollini, pers. observ.). We are therefore limiting the scope of this paper to the southern range of Northeastern Bulrush. To date, we have done most of the existing ecological research on this species, particularly in the southern range of Northeastern Bulrush, focusing on factors that affect germination, survival, growth, and distribution (e.g., Lentz 1998, 1999; Lentz and Cipollini 1998; Lentz and Dunson 1999; Lentz and Johnson 1998; Lentz-Cipollini and Dunson 2006).

Important threats to Northeastern Bulrush include loss or alteration of the temporary wetland habitats on which it depends due to hydrologic modification, fragmentation, fire suppression, logging, mining, and other forest uses. About half of the extant populations are located on public land, affording them some level of protection. Isolated wetland habitats tend to have a higher proportion of rare species than non-isolated wetlands (Hérault and Theon 2008), most likely due to difficulty in effective dispersal in fragmented habitats (Ozinga et al. 2009). Since Northeastern Bulrush responds to hydrology (Lentz and Dunson 1998, Lentz-Cipollini and Dunson 2006), global warming may also pose a particular threat to this species through alteration of hydrologic regimes (Bauder 2005, Brooks 2009). Although Northeastern Bulrush suffers little from insect herbivores and disease, it is sensitive to simulated vertebrate herbivory (Lentz and Cipollini 1998), and an important biotic threat is grazing by *Odocoileus virginianus* (Zimmermann) (White-tailed Deer; hereafter

"Deer"). Wetland and aquatic plant invaders also have the potential to severely impact this species, due to its restriction to small, isolated wetland habitats. Finally, it is thought that shifts in forest species composition from *Quercus* spp. (oaks) to *Acer* spp. (maples) (due to a number of factors including changes in fire regime, an impact not only of forest management but also of climate change) will affect understory light conditions (Abrahamson and Gohn 2004), which will in turn adversely affect this relatively high-light-requiring species (Lentz and Cipollini 1998, Lentz and Dunson 1999).

To date, monitoring, research, and management efforts for this species are fairly limited, leaning towards a "hands-off" protective approach to conservation. On Pennsylvania State Forest land, managed by the Pennsylvania Department of Conservation and Natural Resources (PADCNR), "public plant sanctuaries" have been identified. Of the 52 sanctuaries identified in 2001, 14 of them (27%) are targeted for enhanced protection primarily due to the presence of Northeastern Bulrush. Land managers essentially set the public plant sanctuaries aside, being careful to limit forest management and other activities near sensitive areas, without any active conservation management. Land managers generally lack scientifically based guidelines for monitoring and management for this species. This lack of information is not unexpected, given that, while plants comprise over half of the species listed under the Endangered Species Act, they are allocated less than 5% of federal funding (Roberson 2002).

New populations of Northeastern Bulrush have been discovered since its listing in 1991, which in part prompted the recent recommendation of changing its status from endangered to threatened (USFWS 2009). The discovery of new populations is one factor that can lead to downlisting of a species (Gordon et al. 1997), though complete delisting is generally uncommon (Doremus and Pagel 2001). However, it is clear that this species is not necessarily secure. Many populations lacked detailed status information (C. Copeyon, USFWS, State College, PA, pers. comm.). From a brief survey in 2006, we found that many extant previously studied populations (Lentz 1998) were declining. This finding illustrated the need to revisit existing populations to document their status, to assess a suite of habitat variables, and to document potential threats to each population.

Methods

Using data provided by state natural heritage programs, National Wetland Inventory (NWI) topographic maps, and aerial photos from Google Earth, and aided by a handheld GPS (Earthmate GPS PN-20, DeLorme, Yarmouth, ME), we navigated to each wetland previously known to hold Northeastern Bulrush (Fig. 1). Sites in Pennsylvania were located on State forest land and State game land, with one site on land owned by The Nature Conservancy (TNC). In Maryland, Virginia, and West Virginia, we visited 3 sites managed by the US Forest Service (USFS), 1 site managed by Virginia Department of Natural Resources (VADNR), and 5 privately owned sites. For this study, we visited 90 separate wetlands found at 57 different sites (as this species can be found in multiple wetlands within a site), representing 69% of sites found in Pennsylvania, 70% of sites found in the southern range of Northeastern Bulrush, and 58% of sites range-wide (Table 1). We visited the majority of Pennsylvania sites in July 2008,

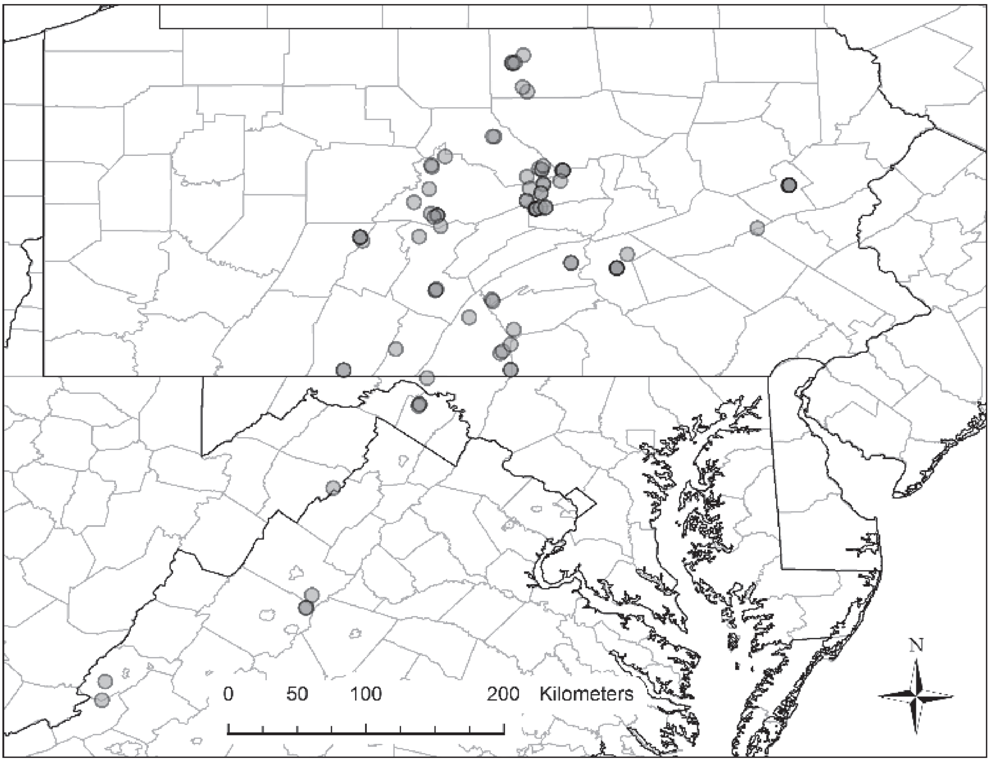

Figure 1. Location of Northeastern Bulrush sites in Maryland, Pennsylvania, Virginia, and West Virginia surveyed for habitat variables, population variables, and threats. Darker circles indicate overlapping wetland points as a result of multiple wetlands per site or as a result of map resolution.

Table 1. Number of extant sites with Northeastern Bulrush (from USFWS 2009 except where noted), number of sites surveyed, and percentage of sites surveyed by state and region.

State	Total number of extant sites	Number of sites surveyed	Percentage of sites surveyed
MD	1	1	100%
PA	70	48	69%
VA	7	5	72%
WV	3	3	100%
NH, VT, MA	41[A]	14[B]	34%
Total across range	122	71	58%
Total in southern region	81	57	70%

[A] Number of sites in NH, VT, and MA are based on current information from R. Popp of VT Fish and Wildlife Department.
[B] Sites surveyed in northern range (i.e., NH, VT, and MA) are not included in current analyses.

but a few sites were visited in October 2007 and October 2008. Sites in Maryland, Virginia, and West Virginia were surveyed in June 2010.

At each wetland, we recorded the coordinates and elevation using the GPS, later checking for accuracy on Google Earth and on USGS topographic maps. We used a fiberglass measuring tape to measure the approximate width and length of the wetland in meters. Boundaries were fairly easy to estimate as there was generally a topographic drop-off and/or a sharp change to forest at the boundary of the wetland. We calculated elliptical wetland area in m^2 by multiplying the half-length and half-width by π. We noted the identity of the mature tree species in the forest within ≈10 m of the wetland and the presence of any recognized wetland or aquatic plant invaders. We also recorded herbivory or damage by Deer on Northeastern Bulrush (which was readily attributable to Deer based on the feeding style and presence of other circumstantial evidence). We recorded instances of other direct threats to the wetland, including evidence of adjacent road drainage and *Ursus americanus* Pallus (Black Bear) wallowing activity.

In the center of each wetland, we used a convex spherical densiometer (Forest Densiometers, Bartlesville, OK) to estimate forest canopy closure in each cardinal direction (N, S, E, and W). The four measurements were averaged to determine percentage of tree canopy closure for each wetland, using the instructions provided on the spherical densiometer. Measuring tree canopy closure in the center of the wetland provides an estimate of the general light conditions of the entire wetland, as the center of the wetland is generally open, with the tree canopy overhanging the wetland from the forest edge. As the forest matures, the canopy gap over the center of the wetland gradually closes, reducing light in the wetland (Fig. 2). Measurements with spherical densiometers may be biased,

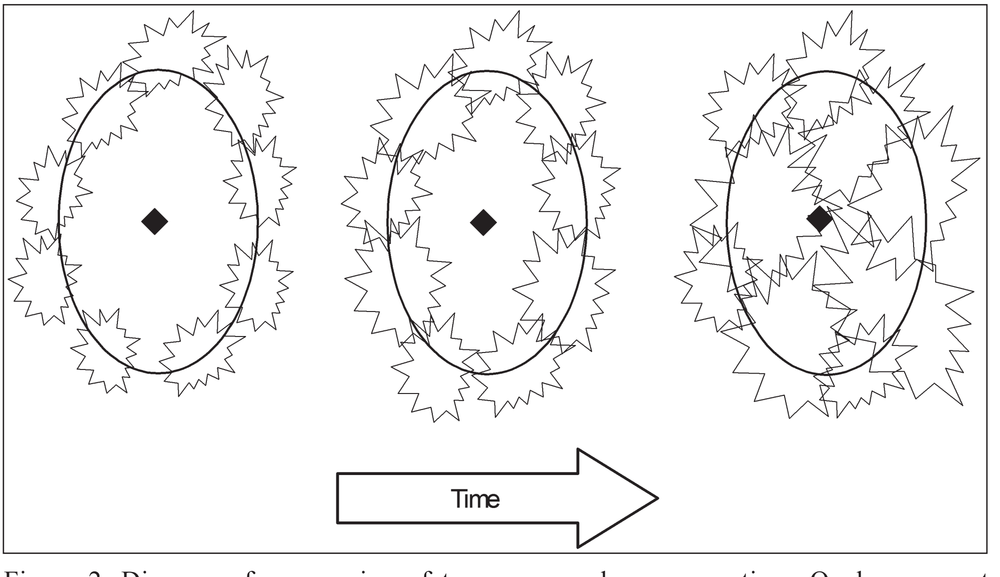

Figure 2. Diagram of progression of tree canopy closure over time. Ovals represent wetland boundaries, irregular polygons represent tree canopy, and solid black diamonds represent where tree canopy closure was measured.

yet can be more precise than other methods for measuring vertical canopy cover (Cook et al. 1995). The densiometer is better suited to measuring canopy closure, i.e., the "proportion of the sky hemisphere obscured by vegetation when viewed by a single point," rather than canopy cover, i.e., the "area of ground covered by a vertical projection of the canopy" (Jennings et al. 1999). Nuttle (1997) argues that angular methods such as the spherical densiometer may be a better assessment of an organism's perception of cover. The spherical densiometer represents a tradeoff between speed of measurement and accuracy (Korhonen et al. 2006). The measurements taken by a spherical densiometer can also later be converted to percent canopy cover if desired by developing predictive models specific to a given ecosystem (Fiala et al. 2006). Taking all of these factors into consideration, the spherical densiometer can easily provide precise comparative measurements of the light conditions experienced by Northeastern Bulrush at each wetland. Canopy closure was not measured in October, at which time tree leaves were already beginning to fall.

In many wetlands (generally towards the center), there was one large fairly uniform monoculture of Northeastern Bulrush, as is common for asexually reproducing species. We measured the approximate length and width of the patch (or patches) of Northeastern Bulrush and calculated the areal extent of the population in each wetland by multiplying the width of each patch by its length. In each patch of Northeastern Bulrush, we counted the total number of stems and the number of flowering stems of Northeastern Bulrush in three 0.25-m^2 areas that appeared to visually represent the average density of the patch. We then averaged the three measurements to find the average density of stems and flowering stems. By multiplying the average densities by areal extent, we estimated the total number of stems and flowering stems in each wetland. We used number of stems rather than number of individuals since determining number of individuals in this clonal species is not possible in the field. To control for possible wetland size effects, we also calculated the percentage area of each wetland occupied by Northeastern Bulrush. In wetlands where populations were small, the number of stems and number of flowering stems in the entire population were counted directly. We performed pairwise correlations to examine relationships among six habitat and Northeastern Bulrush variables: wetland area, percentage of wetland occupied by Northeastern Bulrush, total number of stems, density of stems, percentage of flowering stems, and percentage of tree canopy closure (Ryan et al. 2005).

Based on our observations, we developed a comparative element occurrence (EO) ranking system for this species, based on number of stems, metapopulation structure, threat assessment, population change, and qualitative assessment of habitat. Element occurrence rankings for each site were based in part on guidelines of NatureServe (Hammerson et al. 2008). Due to year-to-year variation in number of stems, it is difficult to rank sites solely on the number of stems. Additionally, number of stems does not necessarily represent the number of individuals, and thus, genetic diversity may be low even if number of stems is high. We therefore developed a ranking for each site that took into account not only the number of stems, but also the threats, recent changes in stem number, and landscape context. Those sites with a high and stable number of stems, low threats, and nearby appropriate habitat that provided dispersal opportunity were ranked highest. Rankings proceed downward from A (the best condition) through F (the worst condition). When populations were intermediate between rankings, they received a two-letter ranking. For 59 wetlands, there was enough information on population size and stem number from previous surveys, either in the Natural Heritage databases or from our own site visits, to make a comparative qualitative evaluation of the status of the population, similar to qualitative assessments found in USFWS (2009). We categorized populations as increased (≈25% increase or greater), stable, decreased (≈25–50% decrease), or decreased greatly (>50% decrease)/locally extirpated. We used rather large thresholds for determining these categories in order to incorporate the fact that some amount of year-to-year variation in population size is expected in Northeastern Bulrush.

Results

Wetlands containing Northeastern Bulrush were found at elevations between 225 and 1087 m, with a median of 510 m. Wetlands tended to be small, ranging from 70 m^2 to 5655 m^2, with a median of 481 m^2 (Table 2). The percentage of tree canopy closure had fairly strong relationships with population parameters of Northeastern Bulrush (Table 3). Percentage of tree canopy closure was negatively correlated with percentage of wetland occupied by Northeastern Bulrush, total number of stems, stem density (Fig. 3), and percentage of flowering stems. Wetland area was negatively related to the and percentage of tree canopy closure and positively related to the percentage of flowering stems, with a tendency to be positively related to stem density.

The invasive plant species *Phalaris arundinacea* L. (Reed Canarygrass) was present at 7% of wetlands, and was the probable cause of extirpation at one heavily invaded site, while invasive *Microstegium vimineum* (Trin.) A. Camus (Japanese Stiltgrass) was present at 21% of wetlands. Deer activity (either trampling or

Table 2. Sample size, minimum, maximum and median values for wetland habitat measures and population variables of Northeastern Bulrush, for the subset of wetlands containing Northeastern Bulrush.

Measure	*n*	Minimum	Maximum	Median
Wetland area (in m^2)	82	70	5655	481
Elevation (in m)	82	225	1087	510
Areal extent (in m^2)	82	1	1236	3
Percentage of wetland occupied	78	<1	100	61
Total number of stems	82	2	116,971	294
Stem density (in number of stems/m^2)	38	21	137	63
Percentage of flowering stems	83	0	97	19
Percentage of tree canopy closure	80	0	99	71

browsing) was noted in 38% of wetlands, with more significant impacts during the fall. Black Bear activity, including wallows, were observed in 17% of wetlands. In 27% of wetlands, we observed habitat and/or hydrologic modification, such as road drainage discharging directly into wetlands and roads crossing parts of a wetland. Of the 59 wetlands with previous data, 14% had increased populations of Northeastern Bulrush, 34% had stable populations, 25% had decreasing populations, and 27% had severely decreased or locally extirpated populations. Element occurrence rankings developed for this species are described in Table 4. For site EO rankings, 47.4% of sites were ranked "C" or above (implying likely long-term persistence), 40.3% of sites were ranked "CD" or "D", and 12.3% were ranked as "DF" or "F", or species likely absent.

Table 3. Pairwise correlation matrix for Northeastern Bulrush habitat and population variables. Superscripts: $^A = 0.10 < P < .05$, $^B = P < 0.05$, $^C = P \leq 0.005$. Number in parentheses is n, the number of points in each relationship.

	Wetland area	% of wetland occupied	Total number of stems	Stem density	% of flowering stems
% of wetland occupied	-0.065 (78)				
Total number of stems	-0.021 (82)	0.777^C (78)			
Stem density	0.293^A (37)	0.129 (36)	0.341^B (37)		
% of flowering stems	0.341^C (79)	0.096 (78)	0.125 (82)	0.496^C (38)	
% of tree canopy closure	-0.498^C (78)	-0.318^C (76)	-0.488^C (80)	-0.523^C (38)	-0.454^C (80)

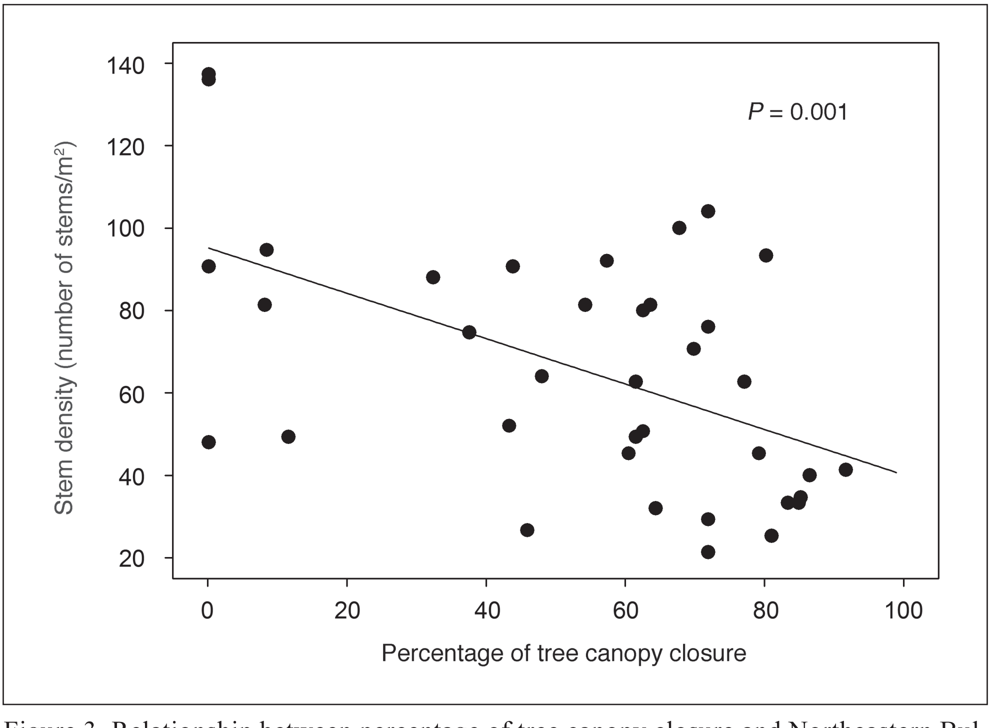

Figure 3. Relationship between percentage of tree canopy closure and Northeastern Bulrush stem density.

Discussion

We visited 90 wetlands known to contain populations of the federally endangered Northeastern Bulrush to document their status, to assess a suite of habitat variables, and to document potential threats to each population. Earlier studies focused on a much smaller scale, studying only 3, 4, or 17 wetlands containing Northeastern Bulrush (Bartgis 1992, Lentz and Dunson 1999, and Lentz-Cipollini and Dunson 2006, respectively). Despite the fact that most of the populations surveyed were found on relatively protected public or conservation lands, over 50% were in decline or possibly extirpated. This finding indicates that the current conservation strategy of setting aside and simply conserving areas with Northeastern Bulrush may be ineffective. Admittedly, populations in ephemeral habitats may undergo population fluctuations (Lesica 1992), but different species do have individualistic responses (Deil 2005). Indeed, Lentz-Cipollini and Dunson (2006) found evidence that population size fluctuates with precipitation input. We would, however, expect that fluctuation in population size would be less severe in general for large populations of this perennial species (capable of both asexual and sexual reproduction), particularly in its southern range, where hydrological fluctuations are presumed to be less severe than in its northern range.

Table 4. Element occurrence (EO) ranking for Northeastern Bulrush.

EO rank description	Letter rank	No. of sites
Population thriving with >15,000 stems in general, excellent example of habitat, prospects for long-term (≈25 yrs) persistence excellent given current condition, intact hydrology and wetland well buffered from development, few to no threats, ample opportunities for dispersal/metapopulation dynamics.	A	2
	AB	3
Population stable or in good condition with >5000 stems in general, good example of habitat, prospects for long-term persistence good given current conditions, hydrology largely intact and wetland mostly well buffered from development, some threats, little opportunity for dispersal/metapopulation dynamics.	B	3
	BC	9
Population declining or condition only fair with >500 stems in general, fair example of habitat, hydrology somewhat compromised with minimal buffer, obvious threats, prospects for long-term persistence uncertain (but still likely), little opportunity for dispersal/metapopulation dynamics, management necessary within next 5 years.	C	10
	CD	10
Population very small (<500 stems in general), degraded habitat, hydrology and buffer clearly compromised, obvious threats, high probability of extirpation if current conditions continue, little to no opportunity for dispersal/metapopulation dynamics, management necessary immediately.	D	13
	DF	1
Population not found, degraded habitat, obvious threats, and most likely locally extirpated under current conditions.	F	6

Current recommendations for forest management adjacent to wetlands includes a no-cut buffer (C. Firestone, PADCNR, Wellsboro, PA, pers. comm.), which may actually be detrimental to this relatively high-light requiring species. Our study is the first to provide replicated data correlating forest canopy closure over wetlands with population parameters of Northeastern Bulrush on a large scale. Our findings are not unexpected given the known experimental response of this species to light availability (Lentz and Cipollini 1998), and that species composition in isolated forested wetlands can be determined in part by plant light requirements (Hérault and Theon 2008). Further, the forest canopy closure was negatively related to *percentage* of wetland occupied by Northeastern Bulrush, a variable that removes any confounding effect of wetland size. It is important to note that the data from this study simply provides a snapshot of current conditions and relationships between variables; therefore, it does not experimentally illustrate cause-and-effect. However, for the 17 sites for which we have data from 1994, percentage of tree canopy closure has generally increased by ≈25% overall, with a concomitant decline in populations. Other monitoring efforts have also documented increases in population size with both experimental and natural removal of tree canopy adjacent to wetlands (K. O'Malley, WV Department of Natural Resources, Romney, WV, pers. comm.). Based on this information and on our current results negatively linking percentage of forest canopy with several population parameters of Northeastern Bulrush, we suggest that experimental reduction of tree canopy closure is advisable to adaptively manage the populations and to provide a buffer against other environmental changes. Our recommendations have already been incorporated into the five-year review of the status of this species (USFWS 2009).

Removing a portion of the forest canopy will not only allow more light into the site, but may also have slight effects on the hydrologic regime by changing evapotranspiration rates (Brooks 2005, 2009). Standing water has been observed to be more frequent in cut forests (Russell et al. 2002); however, we are suggesting low levels of canopy removal primarily in an area immediately adjacent to the wetland, and hydrologic effects are therefore expected to be limited. Increased light and any increased water should benefit Northeastern Bulrush (Lentz and Cipollini 1998 and Lentz-Cipollini and Dunson 2006, respectively) provided the water level is not too high (Lentz and Dunson 1998). Higher light levels can also help this species tolerate other forms of stress, such as Deer herbivory (Lentz and Cipollini 1998).

We recommend that the forest canopy for all sites with greater than 70% closure be reduced to 40–50% closure (measured in the center of the wetland using a spherical densiometer) by trimming, girdling, or otherwise killing selected trees on the perimeter of the wetland. We selected this level based on thresholds noted for Northeastern Bulrush occurrence (Lentz and Dunson 1999) as well as our current findings. Populations of Northeastern Bulrush with greater than 70% canopy closure were small, with a low percentage of flowering stems. Reducing forest canopy closure to 40–50% is still within the natural range of variation, yet will allow for more infrequent management events. By carefully tracking how populations change with changing tree canopy closure, with each researcher using the same measurement protocol, the recommended management level can be adjusted as adaptive management warrants. Common trees surrounding and shading these wetlands include *Acer rubrum* L. (Red Maple), *Quercus rubra* L. (Red Oak), *Pinus strobus* L. (White Pine), and *Nyssa sylvatica* Marsh (Black Gum). Which tree species are cut is not particularly important, so land managers can make this decision. For the first trials of using this management method, we recommend the installation of a surface monitoring well and a data logger which measures water level. A continuous water level monitor is necessary as wetlands tend to experience a great deal of variation in water level in short time frames (Lentz 1998). An adjacent unmanaged wetland as similar as possible to the managed wetland should be used for comparative purposes. Subsequent monitoring of the hydrologic regime and population response in both wetlands should occur for 5–7 years. Following the population for a longer time frame is necessary to get a more accurate assessment of the response of this perennial species, which is known to have population variance from year to year. The efficacy of the tree canopy thinning can be evaluated by comparing the population response of Northeastern Bulrush and the hydrologic responses of the experimental wetland to the control wetland. Ideally, other species dependent on this habitat (e.g., amphibians) should also be monitored to assure that the tree canopy thinning treatment does not adversely impact other components of biodiversity of these habitats. In fact, some amphibians may even do better if standing water increases in each wetland (see Russell et al. 2002). Amphibian diversity can actually increase with a decrease in forest canopy closure (Skelly et al. 2005)

Monitoring the presence or absence of threats and summarizing these data across multiple sites can give a quantitative assessment of the potential of each threat for a species of concern (Wixted and McGraw 2009). We observed human habitat and/or hydrologic alteration in nearly one-third of wetlands. Since habitat modification therefore is a fairly common threat and Northeastern Bulrush is sensitive to water levels (Lentz and Dunson 1998) and to changes in natural hydrology (Lentz-Cipollini and Dunson 2006), we recommend more field research into the long-term impact of these anthropogenic activities on Northeastern Bulrush. Deer activity was noted in 38% of the wetlands that we sampled, and ranged from substantial grazing to trampling and other forms of disturbance. Deer often use wetlands as watering holes, and many of the isolated wetlands where Northeastern Bulrush occurs are the only water sources to be found in large tracts of forest. Among the plant species that exist in these wetlands, Northeastern Bulrush also appears to be a preferred species for Deer (D. Cipollini, pers. observ.), especially in fall when it is among the last of the green herbaceous plants in temperate forests. It can tolerate a single bout of simulated Deer herbivory, but low light levels inhibit compensatory ability (Lentz and Cipollini 1998). Restriction of animal activity by fencing may be warranted at wetlands in areas of high Deer densities. Likewise, several wetlands showed evidence of visitation by Black Bear, which often use forested wetlands as wallowing areas. One small population of Northeastern Bullrush appeared to be completely extirpated by chronic wallowing activity. On the other hand, these forms of animal disturbance may be important to create open water and soil sites for seed germination for Northeastern Bulrush, and large animals may be important dispersers of the barbed achenes that Northeastern Bulrush produces, as has been shown for other species with similar seeds (Carter 1993).

Invasive species are generally a more local threat to this species. Eradication of Reed Canarygrass is recommended at the three sites where it was found. Populations of Northeastern Bulrush either in small wetlands or restricted to small areas in larger wetlands seem especially vulnerable, since Reed Canarygrass can readily dominate such areas. A dramatic increase in Reed Canarygrass at one site in Clinton County, PA is most likely the cause of the extirpation of a formerly small population of Northeastern Bulrush that existed at this site. In such instances, complete eradication rather than control should be the strategy (Mack and Foster 2009), which is currently feasible at sites where the invasive population size is small. Another invasive plant, Japanese Stiltgrass, was found co-occurring in only two wetlands with Northeastern Bulrush, but was found adjacent to 21% of the wetlands that we surveyed. Japanese Stiltgrass prefers mesic soils, but it can occupy the edge of seasonal wetlands. Since Northeastern Bulrush tolerates inundation better than Japanese Stiltgrass (K.A. Cipollini, pers. observ.), the opportunity for negative impacts from this invasive species may be limited. Nevertheless, the possible impacts of Japanese Stiltgrass should be more fully investigated, particularly in drier sites. It might be particularly important at wetland edges, where Northeastern Bulrush seedling establishment likely occurs.

We recommend using our standardized monitoring protocol for monitoring the population status of Northeastern Bulrush. In particular, to standardize measures of population size and status, we recommend using stem number and flowering stem number as opposed to counting clumps of ramets. Counting clumps has been used as a method of assessing population size for this species in the past, but the clumps can vary in the number of ramets that they possess. However, because Northeastern Bulrush is clonal, neither the number of clumps of ramets nor the total number of stems necessarily relate to the number of genets in a population. Indeed, our preliminary genetic studies have shown that within-wetland diversity is generally low (K.A. Cipollini, unpubl. data), indicating that each wetland may support few genets. Thus, the best measures of population status will include estimates of population size and genetic diversity. Additionally, we recommend the use of our EO ranking system for this species to ensure consistency across field surveys. This ranking system could be further refined as additional threats are identified.

There is no information on the conservation genetics of Northeastern Bulrush. If newly discovered populations are genetically homogenous with existing populations, then they represent the identification of no new genetic resources. Until we have an understanding of the genetic diversity of this species at local and regional scales, we will not know the extent to which fluctuations in population sizes due to environmental impacts such as climate change will negatively impact the conservation of genetic resources in the field. Even a small loss of population size can reduce genetic resources; for example, a 5% loss in population size from small, isolated populations, caused a 30% decline in genetic differentiation (Butcher et al. 2009). In line with recommendations from the five-year review (USFWS 2009), we are currently working to determine the population genetic structure of Northeastern Bulrush in order to add this important information to our population assessments.

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