Restoration Notes

Restoration Notes have been a distinguishing feature of *Ecological Restoration* for more than 25 years. This section is geared toward introducing innovative research, tools, technologies, programs, and ideas, as well as providing short-term research results and updates on ongoing efforts. Please direct submissions and inquiries to the editorial staff (mingram@wisc.edu and cmreyes@wisc.edu).

Holes: A Novel Method for Promoting Vegetation Restoration (Macao)

Shao-Lin Peng (State Key Laboratory of Biocontrol/School of Life Sciences, Sun Yat-Sen University, Guangzhou 510275, China, +86 20 84039571, lsspsl@mail.sysu.edu.cn), Yi-Na Yu (College of Forestry, South China Agricultural University, Guangzhou 510642, China; State Key Laboratory of Biocontrol/School of Life Sciences, Sun Yat-Sen University, Guangzhou 510275, China), Yu-Ping Hou (State Key Laboratory of Biocontrol/School of Life Sciences; Sun Yat-Sen University, Guangzhou 510275, China), Long-Sheng Yu (State Key Laboratory of Biocontrol/School of Life Sciences; Sun Yat-Sen University, Guangzhou 510275, China), Bao-Ming Chen (State Key Laboratory of Biocontrol/School of Life Sciences; Sun Yat-Sen University, Guangzhou 510275, China), Guan-Feng Liang (Department of Gardens & Green Areas, Macao Civic and Municipal Affairs Bureau, CP3054, Macao SAR, China)

There is a seasonal shortage of water in the tropical and subtropical regions of China, though they have high annual precipitation (nearly 2,000 mm per year). The rainy season (April to September) accounts for 80 percent of the annual precipitation. The soil fertility of ecosystems degraded by deforestation is very poor because of the severe erosion resulting from rain pulses. The seasonal water shortage and soil nutrient loss are the main problems in forest restoration in tropical and subtropical China.

We conducted an experiment with digging holes in a reforestation area to determine if it is possible to harvest rainwater during the rainy season to mitigate the seasonal water shortage. Besides conserving water, the holes may accumulate more litter, which may decompose faster because of the conserved water. The released nutrients may infiltrate the soil, which favors tree growth. Therefore, we expected that dug holes might improve long-term water availability, ameliorate soil conditions, and consequently enhance forest productivity and forest restoration. The specific objective of this research was to investigate this method for further utilization and to provide principles and strategies for drought-resistant vegetation restoration.

Ecological Restoration Vol. 27, No. 1, 2009
ISSN 1522-4740 E-ISSN 1543-4079
©2009 by the Board of Regents of the University of Wisconsin System.

Rainwater gathering has been applied in agricultural production in arid areas. Rainwater harvesting based on the collection and concentration of surface runoff for cultivation has been practiced in different parts of the world. For example, supplemental irrigation with water collected into small ponds is used to increase water availability of crops for improving and stabilizing agricultural production in semiarid regions of China (Xiao and Wang 2003).

We conducted the experiments in Macao, located to the west of the Pearl River Delta on the southeastern coast of China. Macao (26.8 km²) consists of the Macao Peninsula, Taipa Island, and Coloane Island. The annual mean temperature was 22.3°C; the lowest and highest monthly average temperature were 10°C (February 1968) and 29.8°C (August 1990). Average relative humidity was 80%, and annual average precipitation was about 2,103 mm (1996–2005 data) with severe seasonal aridity from October to March.

The main ecosystem in Macao was evergreen broadleaf forest in the 1880s. In the twentieth century, the forests were degraded by human disturbances, such as timber harvesting, litter removal, and natural accidents such as wildfire. In the 1930s, Taipa Island and Coloane Island became bare soil. After the 1960s, reforestation was carried out in Macao. In the bald ground, some pioneer species were planted, such as Masson pine (*Pinus massoniana*), Taiwan acacia (*Acacia confusa*), longleaf beefwood (*Casuarina equisetifolia*), and cinnamon (*Cinnamomum*

Figure 1. The hole used in forest restoration. Note the litter accumulation. Photo by Shao-Lin Peng

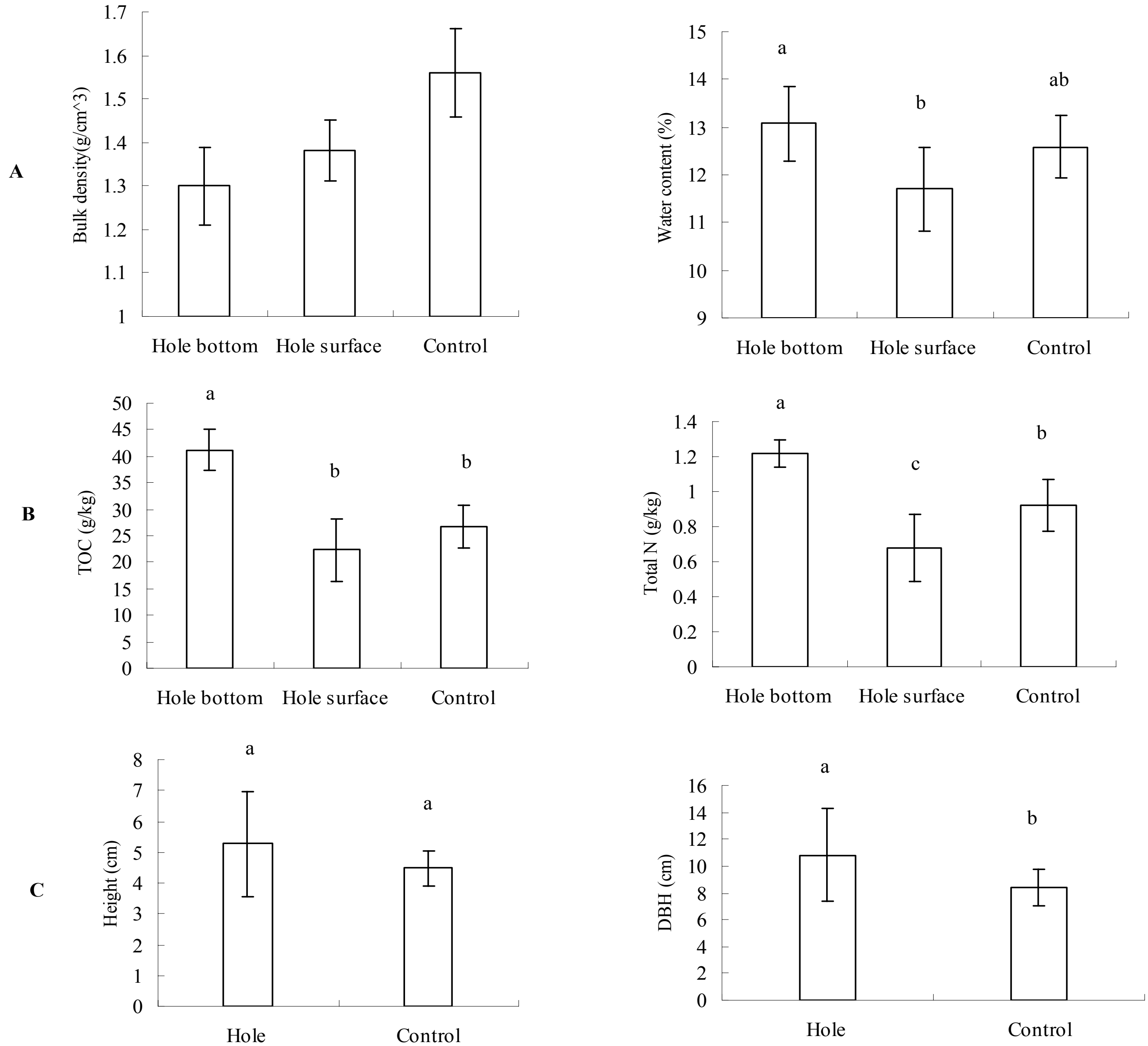

Figure 2. The mean (± SD) effects of holes on soil properties and plant performance: A) Top 20 cm soil physical characteristics; B) Soil chemical characteristics (TOC represents total organic carbon); and C) Average height and diameter at breast height (DBH) of Taiwan acacia (*Acacia confusa*) in different locations. "Hole bottom soil" was sampled in the bottom of the holes; "hole surface soil" was taken up to 20 cm distance away; "control soil" was taken over 200 cm away from the holes. "Hole" was Taiwan acacia within 20 cm of the holes in the forest; "control" was Taiwan acacia over 200 cm away from the holes. Lower-case letters represent significant difference at α = 0.05.

sp.). When the soil and environmental conditions were improved, some zonal evergreen species were introduced in succession. During the forest restoration, seasonal drought is the limiting factor (Peng et al. 2004).

During the restoration process in Macao in the 1990s, we dug 1 × 0.5 m holes, 0.5–1m deep and more than 10 m apart, in bare ground, especially in cliffy slopes (Figure 1). We collected soil cores (10 cm height, 5.4 cm diameter) in a forest in the Central Mountain of Coloane Island in October 2005. Four replicate "surface soil" samples were taken at distances up to 20 cm distance away from the holes. Simultaneously, four replicate "control" samples were taken over 200 cm away from the holes. Because we observed much larger litter horizons in the holes than on the surface soil, we sampled soils in the base of the holes as "hole bottom soil" to monitor the nutrient enrichment in the holes. The soil samples were brought to the lab and air-dried after collection. Soil subsamples were passed through a 2-mm sieve for physical analysis and a 0.5-mm sieve for chemical analysis. We measured the bulk density (cutting ring method), moisture (gravimetrically 105°C for 24 h), organic matter (potassium dichromate method), and total nitrogen (Kjeltec auto 1030 analyzer, Tecator, Höganäs, Sweden) of each sample. Soil water content is an important factor affecting plant growth, and soil bulk density is an important index of the soil fertility level.

Acacias are pioneer species used in restoration here. Taiwan acacia is one of the dominant species, distributed in every sampling point of this research. The diameter at breast height (DBH) and height (H) of Taiwan acacia in surface soil near the holes and in control soils were measured.

The soil bulk density of the control treatment was the highest (independent t-test, $p < 0.05$), and there was no significant difference between surface and hole bottom samples (Figure 2A). The soil water content of the hole bottom was significantly higher than that of the surface soil ($p < 0.05$, Figure 2A). Total organic matter (TOC) and total N of the hole bottom were significantly higher than those of the hole surface and control ($p < 0.05$, Figure 2B). The average heights and DBHs of Taiwan acacia were greater by the holes (Figure 2C).

It is clear that the growing conditions were better by the holes. The improvement of physical and chemical characteristics could be the main reasons. The increase of TOC and total N indicated that the soil fertility was improved. The water collected from surface runoff may increase soil water content at the hole bottom and increase water absorption of plant roots.

Our results confirm that hole digging is an effective method for vegetation restoration that could be used and popularized in ecological restoration of arid and semiarid areas, especially the seasonally arid area. The key point in successful vegetation restoration is to improve habitat quality. The holes method can control soil erosion, improve soil fertility, improve water content, resist seasonal aridity, and accelerate the course of vegetation restoration.

The tropical region of South China has other areas that have experienced soil degradation and habitat loss, but there is hope, especially when the hole-digging method and other engineering measures are applied in the process of vegetation restoration. Hole depth depends on the forest situation. If the holes are too deep, plant root systems may be damaged, which would then hinder forest restoration. The timing should be considered carefully before taking action as well. It is unreasonable to dig holes during the dry season because soil is too hard and it could lead to more soil water loss.

Acknowledgments

This study was supported by Foundation of Macao Civic and Municipal Affairs Bureau (No. 709/SZVJ/2005), the National Natural Science Foundation of China (No. 30670385), the Key Project of Chinese Ministry of Education (No.704037), and the Scientific Research Fund, Hongda Zhang, Sun Yat-Sen University.

References

Peng, S.L., H.F. Lu and G.F. Liang. 2004. Vegetation restoration in the two sub-islands of Macao and its benefits (in Chinese). *Ecology & Environment* 13:301–305.

Xiao, G. J. and J. Wang. 2003. Research on progress of rainwater harvesting agriculture on loess plateau (in Chinese). *Acta Ecologica Sinica* 23:179–188.

Invasive and Non-native Species in a Small New England Watershed (New York)

Mei-Yin Wu and Dennis Kalma (Center for Earth & Environmental Science, State University of New York College at Plattsburgh, 101 Broad St, Plattsburgh, NY 12901, 518/564-4036, wum@plattsburgh.edu)

This project surveyed the vegetation community of 444 sampling locations within 40 wetlands in a small New England watershed, the Boquet River, in the northern Adirondack Mountains in Essex County, New York, during summer 2005 and 2006. The collected data served as baseline for a newly established long-term monitoring program. The Boquet River is a major tributary to Lake Champlain. Its 280-mi² watershed is situated within the New York State Adirondack Park, the largest publicly protected area in the contiguous United States. We selected the 40 wetlands, ranging from approximately 2 to 60 ha, to obtain data from as many different situations as possible with regard to wetland type, elevation, size, degree of disturbance, and location.

For each wetland, a baseline and multiple transect lines perpendicular to the baseline were chosen nonrandomly to position sampling locations across the wetlands in order to survey as many different vegetation associations as possible. The number of sampling locations per wetland ranged from 7 to 37, depending on the size of the wetland. To ensure all sampling sites were within a wetland, we followed standard wetland delineation procedures (USACE 1987) so that sampling locations had all three wetland indicators: hydrology, hydrophytes, and hydric soil. We used nested quadrats to survey vegetation according to U.S. Army Corps of Engineers (1987) and U.S. Environmental Protection Agency (2002) procedures. Each sampling location might contain up to four strata: tree, woody vine, sapling/shrub, and herb (Table 1).

Table 1. Vegetation quadrat sampling sizes and strata distinction based on the plant height and/or diameter at breast height (dbh).

Stratum	Height Distinction	Quadrat Size
Tree	Woody plant with dbh ≥ 8 cm	250 m²
Woody Vine	Woody climbing plant ≥ 1 m tall	250 m²
Sapling/ Shrub	Woody plant ≥ 1 m tall with dbh < 8 cm	30 m²
Herb	All plants < 1 m tall	1 m²

Each plant species with foliage extending into the quadrat was identified and its percent cover estimated and recorded. The dominant species of each stratum were determined using the 50/20 rule (USACE 1987), that is, those species whose cumulative percent cover (when added from highest to lowest rank) exceeds 50% and any individual plant species whose percent cover is at least 20%. We paid special attention to non-native and invasive species. Species were identified and categorized as alien to the area according to the information in Gleason and Cronquist (1991) and the USDA Plants database.

We identified a total of 549 species, 40 of which were not native to the region (Table 2). Ten species were categorized as invasive by the Adirondack Park Invasive Plant Program (2008). Invasive plants were present in 52 of the 444 sampling locations. Among them, purple loosestrife (*Lythrum salicaria*) was the most abundant, with up to 84 percent cover, and was found at 36 of the sampling locations. Purple loosestrife is considered a dominant species at 12 of the 36 sites. Common buckthorn (*Rhamnus cathartica*) was another aggressive invader and had completely covered at least one stratum at three of the 22 sampling sites that it invaded. In addition to the 10 invasive species, another 30 non-native plants were found at 78 of the 444 sampling locations (Table 2).

The invasive potential of the 30 non-native species not currently on the Adirondack Park Invasive Plant Project's list has not been well documented and remains unclear. However, seven of the 30 non-native plants were dominant species in at least one sampling site and have potential to be strong invaders in the future (Table 2). Those seven are creeping jenny (*Lysimachia nummularia*), climbing nightshade (*Solanum dulcamara*), European bur-reed (*Sparganium emersum*), bird vetch (*Vicia cracca*), showy buttercup (*Ranunculus acris*), narrowleaf cattail (*Typha angustifolia*), and false baby's breath (*Galium mollugo*). We suggest special attention be paid to the distribution and invasive potential of the above-listed seven species.

Table 2. Introduced plant species observed at 40 wetlands within the Boquet River watershed. Shaded plant species were dominant in at least one of 444 sampling sites; those marked with * are categorized as invasive species by the Adirondack Park Invasive Plant Program.

	Scientific Name	Common Name
	Agrostis gigantea	redtop
*	*Alliaria petiolata*	garlic mustard
	Arctium minus	lesser burdock
	Asparagus officinalis	garden asparagus
	Barbarea vulgaris	garden yellowrocket
*	*Centaurea biebersteinii*	spotted knapweed
	Daucus carota	Queen Anne's lace
	Epipactis helleborine	broadleaf helleborine
	Galium mollugo	false baby's breath
	Glechoma hederacea	ground ivy
	Hesperis matronalis	dames rocket
	Hieracium caespitosum	meadow hawkweed
	Hieracium maculatum	spotted hawkweed
	Hypericum perforatum	common St. John's wort
	Inula helenium	elecampane inula
*	*Iris pseudacorus*	yellow flag
	Leonurus cardiaca	common motherwort
	Leucanthemum vulgare	oxeye daisy
*	*Lonicera morrowii*	Morrow's honeysuckle
*	*Lonicera tatarica*	Tartarian honeysuckle
	Lotus corniculatus	birdsfoot trefoil
	Lysimachia nummularia	creeping jenny
*	*Lythrum salicaria*	purple loosestrife
	Myosotis scorpioides	true forget-me-not
*	*Phragmites australis*	common reed
*	*Polygonum cuspidatum*	Japanese knotweed
	Polygonum hydropiper	marshpepper smartweed
	Ranunculus acris var *acris*	showy buttercup
*	*Rhamnus cathartica*	common buckthorn
*	*Salix fragilis*	crack willow
	Saponaria officinalis	bouncingbet
	Silene vulgaris	maidenstears
	Solanum dulcamara	climbing nightshade
	Sparganium emersum	european bur-reed
	Tanacetum vulgare	common tansy
	Trifolium pratense	red clover
	Tussilago farfara	coltsfoot
	Typha angustifolia	narrowleaf cattail
	Verbascum thapsus	common mullein
	Vicia cracca	bird vetch

Acknowledgments

The project was funded by the Wetland Programs Development Grant, U.S. Environmental Protection Agency (CD-97294901-0). We thank the Boquet River Association, New York State Department of Environmental Conservation, the New York State Adirondack Park Agency, the Adirondack Invasive Plant Project and the State University of New York at Plattsburgh. We extend special thanks to all landowners for granting permission to survey wetlands on their land. The field work for this project was accomplished through the dedication of the following Plattsburgh State University students: Jannel Gabriel, Marc Morgenstern, Michael Praeger, Andrew McMillan, and Lou Trotta.

References

Adirondack Park Invasive Plant Program. 2008. Adirondack Park's invasive plant invaders and potential plant threats. www.adkinvasives.com/PlantList.html

Gleason, H.A. and A. Cronquist. 1991 *Manual of Vascular Plants of Northeastern United States and Adjacent Canada*, 2nd ed. Bronx NY: New York Botanical Garden.

U.S. Army Corps of Engineers (USACE). 1987. Wetlands delineation manual. U.S. Army Corps of Engineers Technical Report Y-87-1.

U.S. Environmental Protection Agency (USEPA). 2002. Methods for evaluating wetland conditions: Using vegetation to assess environmental conditions in wetlands. U.S. Environmental Protection Agency Report EPA-822-R-02-020.

Seed Bank Colonization in Tidal Wetlands following *Phragmites* Control (New Jersey)

Kris D. Hallinger (ARCADIS, kris.hallinger@arcadis-us.com) and Joseph K. Shisler (ARCADIS—Cranbury, New Jersey, USA)

We used an innovative recolonization approach for enhancement of tidal wetlands to achieve large cost savings (compared to traditional mitigation) because it eliminated the need to remove large quantities of marsh soil to control common reed (*Phragmites australis*) and the need for plantings. It also allowed natural regrowth and succession of tidal marsh vegetation that will be sustainable over the long term as a result of natural competition. The first step involved selection of an appropriate mitigation site with a monoculture of *Phragmites*. Second, a boundary ditch was constructed to form a hydrologic barrier to inhibit the spread of *Phragmites* rhizomes into the mitigation area from the surrounding marsh. Next, a *Phragmites* control program was implemented that combines the use of Rodeo® (glyphosate) herbicide and cutting/removal of dead *Phragmites* stalks. Finally, a five-year monitoring program documented recolonization from the remnant tidal marsh seed bank and directed follow-up *Phragmites* control and vegetation enhancement activities.

Tidal wetland mitigation was required by U.S. Environmental Protection Agency Region 2 to compensate for wetland disturbance during remediation activities at a Superfund site on the Raritan River in Edison, New Jersey. Estuarine wetlands on this reach of the tidal Raritan River are dominated by dense monocultures of *Phragmites*, with infrequent patches of indigenous low-marsh vegetation (e.g., smooth cordgrass [*Spartina alterniflora*]), typically in lower elevation areas, similar to other coastal areas of New Jersey (Philipp and Field 2005). A number of studies have discussed the restoration of *Phragmites*-dominated systems using herbicide treatment followed by mowing with mulching or fire to remove the standing dead stalks (Fell et al. 1998, Mitsch 2000). Early field observations of spontaneous regrowth of both low-marsh and high-marsh species (e.g., saltmeadow cordgrass [*Spartina patens*] and seashore saltgrass [*Distichlis spicata*]), in *Phragmites* areas disturbed by remediation activities that exposed the muck substrate, confirmed the existence of a remnant seed bank.

As part of the project planning process, we performed an alternatives analysis to identify a viable mitigation method and developed a Conceptual Mitigation Plan involving enhancement of approximately 4 ha of *Phragmites*-dominated tidal marsh adjacent to the river, based on a 3:1 ratio of disturbance to restoration within the approximately 16 ha wetland area. We also prepared a Mitigation Work Plan to guide implementation of the project based on the accepted alternative involving *Phragmites* control or

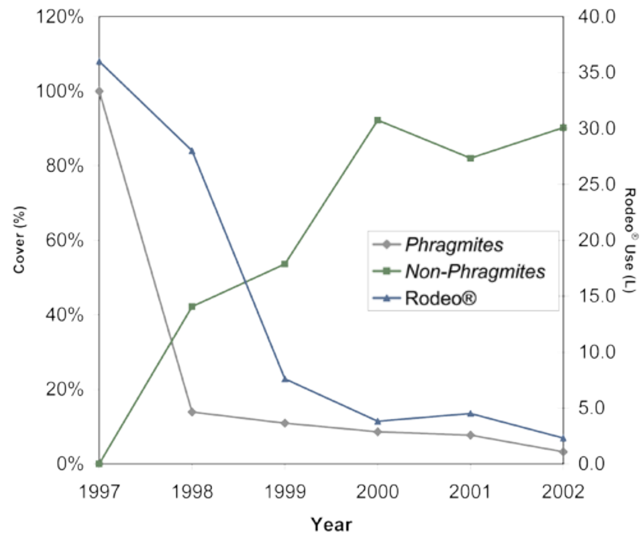

Figure 1. Vegetation coverage and Rodeo® (glyphosate) use in a tidal wetland mitigation project along the Raritan River in Edison, New Jersey, to convert 4 ha of common reed (*Phragmites australis*) to indigenous salt marsh.

removal and recolonization of indigenous species from the remnant seed bank. Mitigation project goals included 1) matching the expected type of wetland vegetation for this area based on an evaluation of historically disturbed and recovered tidal wetlands in an adjacent area and 2) 85% coverage of the mitigation area after five years by an indigenous wetland community.

In 1998 and 1999, we prepared the 4-ha mitigation area by surveying to establish site boundaries, excavating a boundary ditch using a marsh buggy (low ground pressure) excavator, spraying the entire *Phragmites* monoculture within the mitigation area in early fall with Rodeo® (4% a.i. solution) via amphibious vehicles, and cutting and grinding dead *Phragmites* stalks in late winter using a heavy-duty, all-terrain "brush hog" mowing vehicle. These activities resulted in an exposed, unvegetated marsh surface across the mitigation area and created conditions conducive to recolonization of indigenous tidal wetland species from the remnant seed bank.

Following site preparation, we randomly established five transects with a total of 57 quadrats (1 m²) across the mitigation area and measured percent cover for each species annually for five years during the fall growing season. Qualitative vegetation monitoring in the spring identified some additional species, but the majority of these were also present during the fall monitoring event. *Phragmites* cover substantially decreased over time, along with the amount of glyphosate required to treat *Phragmites* during focused annual applications (Figure 1), so that only backpack spraying and no additional cutting was required after Year 2 (1999). The only vegetation enhancement involved interim seeding from onsite seed sources within the mitigation area to enhance seed dispersal into limited low-growth areas near mud flats during Years 2 and 3.

Table 1. Vegetative cover by species in a tidal wetland mitigation project along the Raritan River in Edison, New Jersey, that converted 4 ha of 100% common reed (*Phragmites australis*) to indigenous salt marsh. Dashes indicate absence in quadrats; 0* indicates observed in quadrats at very low frequency and non-quantifiable percent coverage.

Cover Type		% Cover				
Scientific Name	Common Name	1998	1999	2000	2001	2002
Symphyotrichum tenuifolium	Saltmarsh aster	—	—	9.3	5.1	0.1
Atriplex patula	Spear saltbush	0.44	1.3	7.6	0.1	0.0
Baccharis halimifolia	Eastern baccharis	—	—	—	—	0.5
Distichlis spicata	Seashore saltgrass	6.7	2.6	14.1	12.2	17.7
Echinochloa walteri	Coast cockspur	—	—	—	0.3	0*
Eleocharis parvula	Dwarf spikegrass	0*	0.5	0*	0*	1.2
Chamerion angustifolium ssp. angustifolium	Fireweed	—	—	—	—	0.1
Eupatorium sessilifolium	Upland boneset	—	—	—	—	0.4
Iva frutescens	Big leaf sumpweed	0*	0.7	1	1.1	1.5
Mentha spp.	Mint	—	—	—	—	0.7
Phragmites australis	Common reed	13.9	10.9	8.6	7.7	3.2
Pluchea carolinensis	Marsh fleabane	3	2.9	5.2	0.2	2.5
Polygonum hydropiper	Mild water pepper	—	—	—	2.6	0.1
Salicornia maritima	Slender glasswort	1.1	5.7	0*	0.6	0.3
Schoenoplectus americanus	American bulrush	—	—	—	1.9	2.5
S. robustus	Sturdy bulrush	2.6	0.9	7.5	8.5	5.0
Solidago sempervirens	Seaside goldenrod	—	—	—	—	0.4
Spartina alterniflora	Smooth cordgrass	2.8	7.4	12.5	15.8	24.4
S. cynosuroides	Big cordgrass	1.9	2.5	13.3	5.8	4.9
S. patens	Saltmeadow cordgrass	10	18.2	13.1	19.4	24.2
Typha latifolia	Broad-leaf cattail	—	—	—	0.7	0.5
TOTAL		42.4	53.6	92.2	82	90.2

The dense monoculture of *Phragmites* initially was converted to bare ground, which then succeeded to a diverse tidal marsh community with more than 20 indigenous plant species and less than four percent common reed cover (Table 1). Recolonization by indigenous species progressed from Year 1 (1998) onward, with consistent increases in vegetative cover through Year 5 (Table 1). Areas dominated by high-marsh grasses and big cordgrass (*Spartina cynosuroides*) were primarily at elevations greater than 1.2 m above mean sea level (msl), while areas recolonized by smooth cordgrass were typically at elevations less than 1.2 m. Smooth cordgrass dominates the low marsh, and a mixture of saltmeadow cordgrass and seashore saltgrass dominates high-marsh areas. Numerous other desirable indigenous grass and forb species also became established within the mitigation area (Table 1). Early successional or pioneer species, such as saltmarsh fleabane (*Pluchea carolinensis*) and slender glasswort (*Salicornia maritima*), observed in the early stages of recolonization were observed less frequently by 2001. Other species, such as sturdy bulrush (*Schoenoplectus robustus*) and cordgrasses (*Spartina* spp.), became dominant and are indicative of high-quality low-marsh and high-marsh tidal wetland communities. The plant species diversity was greater than expected.

Glyphosate application was highly effective at controlling *Phragmites,* thus allowing indigenous species to recolonize the mitigation area. Additionally, the boundary ditch was highly effective at reducing *Phragmites* rhizome migration into the mitigation area. Minor slumping of ditch banks and banks along the adjacent tidal creek, as a result of the absence of *Phragmites* rhizomes, created intertidal habitat with gentle slopes conducive to smooth cordgrass colonization or mud flat formation. The mitigation area was also opened up to tidal action and recolonization by hermit crabs (*Uca* sp.) in low-lying areas, and unvegetated areas at lower elevations developed into mud flats that were a welcome addition as foraging habitat for wading birds.

Use of the remnant seedbank approach led to recolonization in a mosaic pattern of vegetative coverage by indigenous species and species succession, with early pioneer species giving way to species representative of a salt marsh climax community. Adaptive management in the form of interim seeding from onsite sources may have been useful but unnecessary, given the strength of the seed bank and natural propagation. Mitigation project goals were met each year based on annual monitoring data, and the final goals of 85% vegetation coverage and matching the expected type of wetland system for this area were achieved in fall 2002. The recolonization approach (estimated cost of $62,500/ha) yielded large cost savings compared to traditional mitigation (estimated cost range of $312,000/ha to $1.25 million/ha, Bearmore et al. 2007, Catena et al.

2006), and allowed regrowth and succession of tidal marsh vegetation that should be sustainable over the long term as a result of natural competition. Qualitative observations during the past several years have confirmed the sustainability of diverse vegetation coverage on the mitigation area.

References

Bearmore, B.M., K.A. Donahue and R. Weichenberg. 2007. Partnering for a better saltwater marsh: Design process, construction and monitoring. Poster presented at the 2nd National Conference on Ecosystem Restoration, Kansas City MO, April 23–24.

Catena, J.G., C. Alderson, S. Marseca and J. Sacco. 2006. Fifteen years of restoration activities using oil spill settlement funds in the Hudson-Raritan Estuary. Presentation at the 3rd National Conference on Estuarine and Habitat Restoration. New Orleans LA, December 12.

Fell, P.E., S.P. Weissbach, D.A. Jones, M.A. Fallon, J.A. Zeppieri, E.K. Faison, K.A. Lennon, K.J. Newberry and L.K. Reddington. 1998. Does invasion of oligohaline tidal marshes by reed grass, *Phragmites australis* (Cav.) Tin. Ex Steud. affect the availability of prey sources for the mummichog, *Fundulus heteroclitus* L. *Journal of Experimental Marine Biology & Ecology* 222:59–77.

Mitsch, W.J. 2000. Self-design applied to coastal restoration. Pages 554–564 *in* M.P. Weinstein and D.A. Kreeger (eds), *Concepts and Controversies in Tidal Marsh Ecology*. Norwell MA: Kluwer.

Philipp, K.R. and R.T. Field. 2005. *Phragmites australis* expansion in Delaware Bay salt marshes. *Ecological Engineering* 25:275–291.

Developing an Interdisciplinary Restoration Plan for Napahai Wetland, Yunnan, China

Beth A. Lawrence (University of Wisconsin–Madison, Department of Botany, Madison WI, USA, 608/852-3595, balawrence@wisc.edu), Heqi Wu (Kunming Institute of Zoology, Chinese Academy of Sciences, Kunming, Yunnan P.R. China) and Qiang Liu (Graduate School of the Chinese Academy of Sciences, Beijing, P.R. China)

The restoration of biological diversity is inherently an interdisciplinary challenge, addressing ecological processes that interact with socioeconomic and political forces. This is particularly true in biodiverse regions with developing economies, where local people are frequently dependent on natural resources to maintain their livelihood. The University of Wisconsin–Madison's Biodiversity Conservation and Sustainable Development in Southwest China IGERT (Integrative Graduate Education and Research Traineeship) program brings trainees and Chinese scientists together to address complex, interdisciplinary problems. Here we explore opportunities to restore black-necked crane (*Grus nigricollis*) habitat in Napahai wetland, a high-elevation (3,260 m) marsh located in rapidly changing northwestern Yunnan, China (Figure 1). Approximately 250 of the IUCN-listed vulnerable cranes use the shallow marshes of Napahai from October to May each year (Figure 2). We discuss research that is needed to effectively conserve this species endemic to the Tibetan plateau within this internationally important wetland.

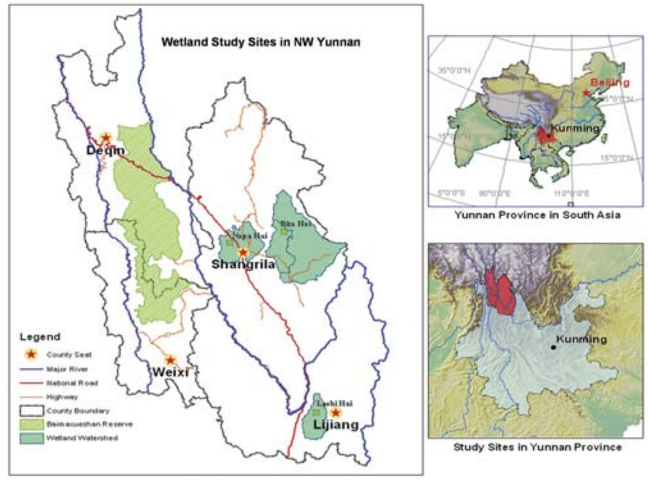

Figure 1. Napahai Wetland is located in northwestern Yunnan province, China, near the city of Shangrila. Map by Rongxun Wang

Napahai Wetland is centered in the core of the Southwest China Biodiversity Hotspot and is an ideal system for conducting interdisciplinary research at the nexus of ecology, livelihoods, and governance. This 2,000-ha wetland is internationally recognized for its role in maintaining hydrologic function at the headwaters of the Yangtze River, and also provides critical habitat for migrating birds, including the revered black-necked crane (Yu 2004). Residents of the 17 surrounding Tibetan villages use the wetland to graze livestock critical to local livelihoods, but unregulated grazing access may have negative consequences on black-necked crane habitat. The growing tourism industry in the region may present both economic opportunities and incentives for locals to restore wetland habitat, but the costs and benefits of different tourism strategies need to be evaluated. Additionally, wildlife habitat restoration plans need to consider recent changes in forest management policies, as land use in surrounding uplands can have dramatic effects on wetland habitat by influencing nutrient and sediment loads.

Increasing tourism and rapid urbanization of Zhongdian, recently renamed Shangrila to entice visitors, may have negative consequences on the biological integrity of Napahai Wetland, but has expanded economic options for locals. Urbanization of Shangrila has increased the need for locally produced pork and vegetables and has possibly resulted in increased use of fertilizer and greater grazing pressure. Local Tibetans residing in villages surrounding Napahai graze an estimated 20,000 head of livestock in the wetland, including pigs, sheep, yak, and horses (Yunnan Forestry Bureau, pers. comm.). Although

Figure 2. The black-necked crane (*Grus nigricollis*) uses Napahai Wetland as wintering grounds in northwestern Yunnan. Note yak in left background, pile of yak dung on right. Photo by Liu Qiang

Tibetans traditionally herded livestock in alpine pastures during summer months for improved forage, the draw of alternative income sources, including ecotourism, makes traditional herding practices difficult to sustain (Wilkes 2006). This increases pressure on lower elevation grazing lands, which are now grazed year-round.

Of particular interest to us is the effect of increasing pig numbers on wetland plant communities and critical bird habitat. Although pigs comprise only a fraction (ca. 5%) of total livestock at Napahai, their rooting behavior is particularly destructive and is considered a serious threat to black-necked cranes. During preliminary reconnaissance, we observed that rooting by Tibetan pigs removes the dominant sedges (*Blysmus sinocompressus*, *Carex pleistogyna*), creating "pig patches" characterized by bare soil and ruderal plant species. While this behavior may be destructive to crane habitat, the vegetation types preferred by black-necked cranes for feeding and roosting habitat are currently unknown.

Research on crane behavioral patterns, habitat use, and feeding preferences could clarify whether pig disturbance is detrimental to crane habitat and whether cranes prefer diverse plant communities. In coming years, researchers from the International Crane Foundation and the Kunming Institute of Zoology are planning to capture several black-necked cranes and equip them with radio collars to monitor their movement throughout the wetland. Spatial and temporal information gained from tracking the cranes combined with detailed vegetation maps will elucidate the habitat preferences of the cranes within Napahai Wetland.

Understanding the specific plant communities that black-necked cranes use and how pig activity affects these communities will clarify how restoration should be implemented. Surveys of plant species distributions and assemblages throughout the wetland, as well as comparisons of plant diversity in disturbed pig patches and matrix plant communities are needed to elucidate how pig patches contribute to vegetation dynamics at Napahai. Evaluation of seed banks to assess the resilience of pig patches and the recolonization rates of pig patches would inform the need for active restoration (e.g.., seeding) to restore plant diversity. Modeling the carrying capacity of cranes versus pigs at Napahai Wetland based on nutritional needs and habitat availability will provide critical and timely guidance for managing this charismatic crane.

Research investigating the efficacy of divergent tourism approaches and their ecological, social, and financial ramifications is necessary to support the sustainable development of Napahai Wetland. The wetland's close proximity to Shangrila (8 km), the newest frontier town in northwestern Yunnan's growing tourist industry, has secured Napahai Wetland as a tourist destination. Droves of tour buses bring hundreds of tourists to Napahai every day to snap pictures of each other in traditional Tibetan garb, ride ponies, and stroll around the wetland. Some consider the extensive horseback riding to be detrimental to wetland habitat, but these activities are generally restricted to small areas near access points. It is currently unclear whether local Tibetans benefit financially from these tourist activities relative to more traditional livelihood practices (i.e., grazing, farming), as tourism concessions negotiated between villages and the local tourism industry may be volatile.

Birdwatching and hiking are emphasized by an ecotourism program developed by the village of Hamagu and the World Wildlife Fund, which also welcomes tourists to stay overnight in Tibetan homes. While this strategy attracts fewer tourists, the villagers are able to charge higher prices for the opportunity to experience traditional Tibetan village life and may also reduce negative ecological and social impacts associated with mass tourism. Hamagu villagers view black-necked cranes as an important resource linked to their economic well-being, but are concerned that unchecked tourism and urban development may destroy remaining crane habitat (Marston 2006). In an effort to protect the wetland and black-necked crane habitat, villagers have joined together to form the Napahai Wetland Community Association. If restoration of wetland habitat is considered necessary and beneficial by local inhabitants, then the likelihood that effective restoration can occur will be greatly enhanced.

In response to massive flooding on the Yangtze and Yellow Rivers, the Chinese government implemented the Natural Forest Protection Program (NFPP) in 1998, which banned logging on over 30 million ha of forest in the upper reaches of these two river systems. Prior to the ban, the logging industry in northwestern Yunnan was a major employer of people from rural communities (Willson 2006), though the logging history of the region is considered to have deleteriously affected biodiversity and forest integrity (Xu and Wilkes 2004). Within the Napahai watershed, commercial logging was likely responsible for slope erosion that resulted in extensive sediment deposition in the wetland.

Some of our preliminary analyses of satellite images have shown that between 1994 and 2000, urban (+58%) and agricultural (+16%) cover increased while forest (-8%) cover decreased, which may be associated with higher nutrient and sediment loads into the wetland, alteration of plant community composition, and ultimately degradation of wildlife habitat in the wetland. Although forest cover declined overall, using spectral mixture analysis we observed an increase in the green vegetation fraction on forested slopes near the city of Shangrila and Napahai Wetland. This suggests that the logging ban is being enforced within the sphere of influence of the Forestry Bureau (i.e., near the city and nature reserve), but may not be strictly enforced farther from institutional centers. These and further land-use analyses will assist in elucidating the drivers of change in the watershed and inform the development of restoration plans. For example, interdisciplinary teams could investigate how the 1998 logging ban has altered local timber harvesting practices, how the urbanization of Shangrila has influenced the livelihoods of villagers surrounding Napahai Wetland and their use of wetland resources, and how locals perceive changes in wetland and water quality over time and their desire and ability to restore ecological function.

Implementation of an effective restoration plan at Napahai needs to account for the interconnected nature of wetland habitat, local livelihoods, and governance. Understanding these relationships will yield context-relevant results that will facilitate the development of sustainable restoration and management strategies. Ultimately, restoration of black-necked crane habitat could enhance the biological integrity of Napahai and promote sustainable livelihoods for those living near this globally important wetland.

Acknowledgments

We would like to thank the University of Wisconsin-Madison Biodiversity Conservation and Sustainable Development IGERT program for funding this preliminary work. We appreciate the guidance and support of Tian Kun, Yang Bo, Yang Xiaojun, Yang Yongping, Josh Posner, and Joy Zedler during this project.

References

Marston, A. 2006. Buddah birds: Protecting the black-necked cranes of Shangri-la. World Wildlife Fund. www.panda.org/about_wwf/what_we_do/species/news/

Wilkes, A. 2006. Innovation to support agropastoralist livelihoods in northwest Yunnan, China. *Mountain Research & Development* 26:209–213.

Willson, A. 2006. Forest conversion and land use change in northwest Yunnan, China. *Mountain Research & Development* 26:227–236.

Xu, J. and A. Wilkes. 2004. Biodiversity impact analysis in northwest Yunnan, southwest China. *Biodiversity & Conservation* 13: 959–983.

Yu, Z. 2004. Information sheet on Ramsar Wetlands (RIS): Napahai Wetland. Gland, Switzerland: Ramsar Convention Bureau. www.wetlands.org/reports/ris/2CN028en.pdf

Bay Scallop Restoration in New York

Stephen T. Tettelbach (Dept of Biology, C.W. Post Campus of Long Island University, Brookville, NY 11548, stephen.tettelbach@liu.edu) and Christopher F. Smith (Marine Program, Cornell Cooperative Extension of Suffolk County, Riverhead, NY 11901, cfs3@cornell.edu)

Natural populations of commercially important bivalve mollusks have declined in many parts of the world during the past century due to overfishing, habitat degradation, disease, and other environmental perturbations. Attempts to augment or restore bivalve populations and fisheries to former levels have benefited from advances made in aquaculture methodology and technology, but these restoration efforts have met with mixed success. One of the most successful programs has sustained the Japanese scallop (*Mizuhopecten yessoensis*) fishery through large-scale production and planting of hatchery-reared animals (Sekino 1992); this is the template for much of the work we describe here.

In the United States, the commercially important bay scallop (*Argopecten irradians*) has declined in most of the Atlantic and Gulf coastal areas where it was abundant in the twentieth century. Bay scallops are particularly susceptible to population fluctuations because they usually live for only 18–22 months and reproduce once. On Long Island, New York, populations of the northern subspecies (*A. irradians irradians*) were nearly extirpated due to direct mortality and recruitment failures during "brown tide" (*Aureococcus anophageferrens*) algal blooms from 1985 to 1987.

Bay scallop restoration in the Peconic Bays (Figure 1) was initiated soon after the first brown tide with the goal of planting juveniles (0+ yr) that would spawn at maturity (ca. 1 yr) and help repopulate the bays. External fertilization is followed by a 1–2 week larval period during which dispersal is effected by tidal currents. Reproductive potential is very high in bay scallops, with a single individual

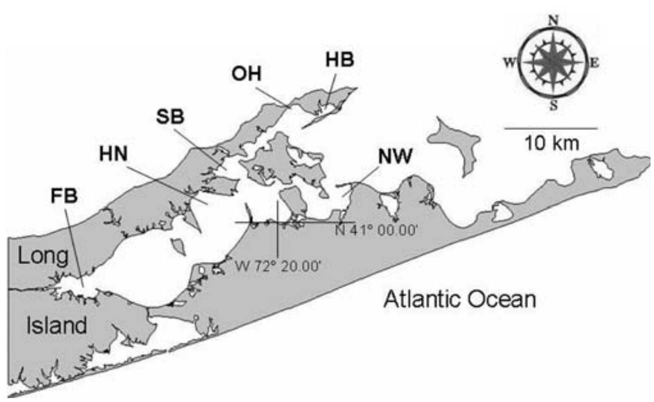

Figure 1. Map of the Peconic Bay system in eastern Long Island, New York, showing the restoration site (Orient Harbor) and other sites monitored for larval settlement to mesh bag spat collectors and/or abundance of juvenile and adult scallops on the bay bottom. OH = Orient Harbor (longline system), HB = Hallock Bay, SB = Southold Bay, HN = Hog Neck Bay, NW = Northwest Harbor, FB = Flanders Bay.

capable of producing over 5 million eggs, so even though survival of transplanted natural and hatchery-reared scallops from the time of planting in the fall until spawning in early summer was less than 12 percent in the late 1980s–early 1990s (Tettelbach and Wenczel 1993), these plantings did contribute to an increase in Peconic Bay scallop populations (Krause 1992).

A severe brown tide in 1995 again decimated populations, but despite the absence of brown tides since then scallop populations have not recovered naturally. Commercial harvests since 1995 have remained at 1%–2 % of pre–brown tide levels (NYSDEC 2008), and population densities in most areas are very low (< 0.1 individuals/m²). We have postulated several explanations, including a possible decline in water quality and the observed loss of eelgrass (*Zostera marina*), the preferred habitat of juvenile and adult bay scallops. However, our primary hypothesis for the lack of recovery is that local bay scallop densities and numbers have been too low to permit high rates of successful fertilization of eggs (Liermann and Hilborn 2001). Experiments with other invertebrates exhibiting external fertilization (such as sea urchins, sea squirts, and hydroids) have shown that fertilization success drops sharply as distance between spawning males and females increases to more than 1 m or population density drops too low (Levitan and Petersen 1995). In turn, larval supply becomes limiting (Peterson et al. 1996). Under these circumstances, populations may not be able to rebuild unless natural mortality rates decline (e.g., by reducing the numbers of predators) or spawning stocks are boosted above some critical threshold.

Most of our early restoration work involved free-planting scallops to the bay bottom by dispersing them by hand from boats that transit back and forth across a demarcated area. These plantings were done at low densities (≤ 10 ind/m²) after Japanese methods and to avoid attracting large numbers of crustacean and gastropod predators.

In our current work, we are attempting to "jump-start" bay scallop populations by planting large numbers of hatchery-reared individuals at high densities to ensure a high probability of fertilization success upon spawning. Our change in approach was prompted by the recent literature on fertilization dynamics (noted above) and our own recent observations of up to 50% survival, from early spring until the initiation of spawning in summer, of scallops free-planted at densities of about 100 ind/m². In our current work we are free-planting at comparable densities, but our major emphasis is deploying scallops in thousands of lantern nets suspended in midwater from subsurface lines—again following Japanese methods. Scallops are thus protected from predators and are concentrated at the time of spawning, since nets are stocked at densities of 190–200 ind/m² and are spaced about 1 m apart on 200+ m long lines. We have stocked from 250,000 to over 500,000 scallops in this system each fall, 2006–2008, at a single site in Orient Harbor (Figure 1), which formerly supported a robust scallop population. Our efforts represent the largest bay scallop restoration project of its kind yet attempted in the United States.

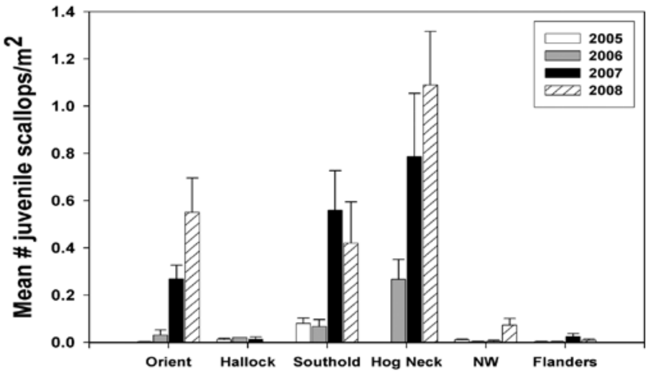

Figure 2. Mean density (± 1 SE) of juvenile (0+ year) scallops in six embayments in eastern Long Island NY during 2005–2008 Orient = Orient Harbor, Hallock = Hallock Bay, Southold = Southold Bay, Hog Neck = Hog Neck Bay, NW = Northwest Harbor, Flanders = Flanders Bay. Data were collected via transect surveys in the fall of each year or spring of the following year (e.g., Fall 2005/Spring 2006).

While survival of our free-planted scallops from fall until the following summer has been low (≤ 7%), the higher survival (36%–50%) of scallops in our lantern nets has yielded many more spawning adults at higher densities than we witnessed in our restoration efforts in the 1980s. Consequently, the numbers of scallop larvae produced by our current stocks should be much higher; this has been confirmed by our 2005–2008 field monitoring.

Levels of larval recruitment to "spat" collectors (mesh bags suspended above the bay bottom), as well as abundance of juvenile and adult scallops on the bottom of Orient Harbor, are markedly higher than in the two years prior to the start of our current restoration work. Larval recruitment at most of our eight monitoring sites in Orient Harbor in 2007 and 2008 increased 3–13 times over respective levels in 2005 and 2006 (data not shown). Overall densities of juvenile (0+ yr) scallops significantly increased (Figure 2, Dunn's test $p < 0.05$, following Kruskal-Wallis test $p < 0.001$), and the spring 2008 estimated juvenile population, based on the area-density method, was 13.5 times higher than that seen in the two years before spawning of our planted scallops. Overall adult scallop densities in fall 2008 were three times higher than in 2005.

In addition to Orient Harbor, we have monitored larval recruitment and/or juvenile scallop densities in other areas where natural population levels were very low (Figure 1), as they were in Orient Harbor in 2005–2006 prior to our plantings; our surveys in these areas (Hallock Bay, Northwest Harbor, Flanders Bay) have shown that juvenile densities have remained at less than 0.1 ind/m² (Figure 2). These results strongly suggest that spawns of scallops in 2007 and 2008, which we planted in 2006 and 2007, respectively, have contributed to the increased population sizes in the Orient Harbor area.

Two other unplanted areas that we have monitored (Southold Bay and Hog Neck Bay) have more than quadrupled in abundance of juvenile scallops since 2005 (Figure 2). At Southold Bay, where we have monitored adult abundance for the longest period, numbers of adults increased sixfold from 2005 to 2008, possibly due in part to larval dispersal from spawning Orient Harbor stock (Siddall et al. 1986). Population densities at Hog Neck Bay may have been high enough since 2006 to sustain themselves.

Results to date suggest that the bay scallop population is increasing in the vicinity of our longline system in Orient Harbor, and we anticipate a measurable increase in bay scallop commercial fishery landings in this area in fall 2008. We have recommended limited area closures to protect juveniles during landings, and we plan to continue our current planting efforts at the same high densities and numbers and are hopeful that these efforts will continue to boost populations toward a level at which they may become self-sustaining, at least 1–2 ind/m² by our estimates.

Acknowledgments

We gratefully acknowledge funding support from the Suffolk County (NY) Water Quality Protection and Restoration Program as well as the Coastal Management Program of New York State Dept. of State. We thank Kevin Cahill, Jeff Chagnon, Mike Patricio, Kim Tetrault, Gregg Rivara, Neal Stark, Josh Clauss, Alli Dinargo, and Ken Birmingham for their tireless work in growing and maintaining bay scallop stocks and the assistance of Bradley Peterson, Scott Hughes, Dennis Bonal, Andrew Weinstock, Chelsea Fitzsimons-Diaz, John Carroll, Jason Havelin, and Richard Ames with field monitoring.

References

Krause, M.K. 1992. Use of genetic markers to evaluate the success of transplanted bay scallops. *Journal of Shellfish Research* 11:199.

Levitan, D.R. and C. Petersen. 1995. Sperm limitation in the sea. *Trends in Ecology & Evolution* 10:228–231.

Liermann, M. and R. Hilborn. 2001. Depensation: Evidence, models and implications. *Fish & Fisheries* 2:33–58.

New York State Department of Environmental Conservation (NYSDEC). 2008. Annual commercial shellfish landings for New York State, 1946–2007. Unpublished report. East Setauket: NewYork State Department of Environmental Conservation.

Peterson, C.H., H.C. Summerson and R.A. Luettich Jr. 1996. Response of bay scallops to spawner transplants: A test of recruitment limitation. *Marine Ecology Progress Series* 102:93–107.

Sekino, T. 1992. Scallop (*Patinopecten yessoensis*). Pages 175–185 *in* T. Ikenoue and T. Kafuku (eds), *Modern Methods of Aquaculture in Japan*, 2nd ed. Tokyo: Elsevier.

Siddall, S.E., M.E. Vieira, E. Gomez-Reyes and D.W. Pritchard. 1986. Numerical model of larval dispersion. Marine Sciences Research Center Special Report No. 71. State University of New York at Stony Brook:.

Tettelbach, S.T. and P. Wenczel. 1993. Reseeding efforts and the status of bay scallop *Argopecten irradians* (Lamarck, 1819) populations in New York following the occurrence of "brown tide" algal blooms. *Journal of Shellfish Research* 12:423–431.

The New Mexico Forest Restoration Principles: Creating a Common Vision

Anne Bradley (The Nature Conservancy in New Mexico, 212 E Marcy St Ste 200, Santa Fe, NM 87501, abradley@tnc.org)

Millions of hectares of ponderosa pine (*Pinus ponderosa*), mixed conifer forests, and piñon-juniper (*Pinus edulis–Juniperus* spp.) woodlands in the southwestern United States are in crisis, owing to past logging, grazing, and fire suppression practices. Historic surface fire regimes have been all but eliminated, and today many areas are dominated by dense stands of small-diameter trees, leaving these forests vulnerable to destructive crown fires. As in many western states, New Mexico's forests are predominantly on federal lands. Decades of controversy over logging practices have left many skeptical of forest management. "People don't trust government . . . and increasingly they don't trust experts" (Cortner 2003). Critics question whether fuel reduction treatments are just traditional timber sales by another name when they remove larger trees and whether creating stands of uniformly spaced trees for fire protection can in any way be considered restorative. Without other avenues to influence management, disagreement has often been expressed through litigation and administrative appeals. Collaborative resource planning with interest groups has been gaining popularity to reduce conflicts in forest management. In this paper, I describe a collaborative group in New Mexico working to accelerate forest restoration at a statewide scale through a set of guiding ecological principles.

Several recent events created conditions supporting collaborative approach to restoration. In 2000, the Cerro Grande Fire severely burned nearly 18,000 ha and 400 homes in the Jemez Mountains of northern New Mexico around Los Alamos, causing the evacuation of 18,000 people. Soon thereafter, the Community Forest Restoration Act of 2000 (Title VI, Public Law 106–393) established the Collaborative Forest Restoration Program, or CFRP (www.fs.fed.us/r3/spf/cfrp/) to manage cost-share grants to private organizations or communities for forest restoration projects on federal, tribal, or state lands. These projects must include diverse stakeholders and address several objectives, including wildfire threat reduction, reestablishment of historic fire regimes, preservation of old and large trees, and increased utilization of small-diameter trees. In the last seven years, over 100 CFRP projects have been funded through a consensus process involving representatives from federal and state agencies, forest industry, local communities, tribal interests, academic research, and environmental groups. Participation in the CFRP has engaged a significant number of people and organizations in the state in on-the-ground projects.

Furthermore, a group of scientists and environmental advocates contributed an influential summary paper (Allen et al. 2002) that proposed a broad framework for ponderosa pine restoration based on ecological principles. This paper was the result of a long debate over different silvicultural practices promoted for ponderosa pine restoration. Then in 2005, the Public Service Company of New Mexico (PNM) began a feasibility study for a 35-megawatt plant fueled by woody biomass. Estimates were that 4,000 ha/year of woody biomass would be needed, and a continuing supply of biomass had to be assured. Project managers brought together representatives from 13 land management agencies, environmental groups, and forest industry organizations, many of whom had participated in the CFRP or contributed to the Allen et al. paper, to consider how biomass removal could take place over a large landscape with minimal conflict. The invited group became the New Mexico Biomass Evaluation Taskforce, which decided that expressing the "zone of agreement" among members as a set of principles would be most effective. The taskforce took the Allen et al. principles and revised and expanded them in a short 18 months (Box 1).

The principles emphasize both the project and landscape scales, as well as effects of management on the understory biodiversity. A noteworthy inclusion is the specific mention of piñon–juniper woodlands, which were targeted for biomass removal. However, scientific information was lacking. The taskforce sought the advice of a number of woodland scientists, leading to a new collaborative group led by Dr. Bill Romme at Colorado State University that published a valuable new summary of the historical and modern disturbance regimes in piñon-juniper systems (Romme et al. 2008). Piñon and juniper have often been treated as grassland invaders, and large landscapes have been cleared of trees. The Romme et al. paper provides a more holistic picture of woodland distribution and dynamics, and the taskforce is developing a framework to help practitioners identify sites as historic woodland, grassland, or savanna to guide restoration.

While the Forest Service is increasing its efforts to collaborate, some internal barriers still exist (Friederici 2003). In our case, the Regional Forester supported the principles, but at the Washington level there was a desire to safeguard the independent decision-making authority of the agency. To ease these concerns, the New Mexico State Forestry Division led an extensive public scoping process. This, and a commitment to standard planning processes, demonstrated that the principles were designed to be a framework for good forestry where restoration was the objective, not an inflexible prescription promoted by a few special interests.

Today, the biomass plant that triggered the New Mexico principles remains in the planning stages. Taskforce partners are still engaged and now are looking for a landscape where they can jointly design a project and test whether their common vision can be translated into a common prescription. Members of the taskforce continue to improve piñon-juniper restoration guidance. As Moote and Lowe (2008) point out in their synthesis of natural resource collaboration, "reaching . . . decisions and outcomes takes considerable effort and time—usually on the order of years." While it is too early for a summary of specific sites where the New Mexico Forest Restoration Principles have prevented conflict, it is clear that they are viewed as helpful. The principles themselves have become common currency in discussing forestry projects in the state: state foresters use them to communicate objectives with private land owners and major environmental groups in the state are active proponents, as are many thinning contractors.

Box 1. The New Mexico Forest Restoration Principles. The complete text of the principles and list of the 13 participating organizations can be found at www.fs.fed.us/r3/spf/nm-restor-principles-122006.pdf.

1. Collaborate.
2. Reduce the threat of unnatural crown fire.
3. Prioritize and strategically target treatment areas.
4. Develop site-specific reference conditions.
5. Use low-impact techniques.
6. Utilize existing forest structure.
7. Restore ecosystem composition.
8. Protect and maintain watershed and soil integrity.
9. Preserve old or large trees while maintaining structural diversity and resilience.
10. Manage to restore historic tree species composition.
11. Integrate process and structure.
12. Control and avoid using exotic species.
13. Foster regional heterogeneity.
14. Protect sensitive communities.
15. Plan for restoration using a landscape perspective that recognizes cumulative effects.
16. Manage grazing.
17. Establish monitoring and research programs and implement adaptive management.
18. Exercise caution and use site-specific knowledge in restoring or managing piñon-juniper ecosystems and other woodlands and savannas.

Other regions have been experimenting with their own principles. I am aware of groups in Colorado, Oregon, and Montana working on collaborative principles. Finally, the U.S. Forest Service (2008) has released a new "Ecological Restoration and Resilience" directive that codifies restoration as a responsibility for Forest Service managers. Taskforce members from the Forest Service point to the New Mexico Restoration Principles as an important influence in the development of this national directive.

Defining a zone of agreement for forest restoration at a statewide scale has created opportunities to develop new scientific syntheses and policies and is helping agencies and their partners articulate restoration objectives across jurisdictional lines.

References

Allen, C.D., M. Savage, D.A. Falk, K.F. Suckling, T.W. Swetnam, T. Schulke, P.B. Stacey, P. Morgan, M. Hoffman and J.T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: A broad perspective. *Ecological Applications* 12:1418–1433. hdl.handle.net/2019/204

Cortner, H.J. 2003. The governance environment: Linking science, citizens and politics. Pages 70–80 *in* P. Friederici (ed), *Ecological Restoration of Southwestern Ponderosa Pine Forests*. Washington DC: Society for Ecological Restoration International and Island Press.

Friederici, P. 2003. The "Flagstaff Model." Pages 7–25 *in* P. Friederici (ed), *Ecological Restoration of Southwestern Ponderosa Pine Forests*. Washington DC: Society for Ecological Restoration International and Island Press.

Moote, M.A. and K. Lowe. 2008. What to expect from collaboration in natural resource management: A research synthesis for practitioners. ERI–Issues in Forest Restoration. Flagstaff AZ: Ecological Restoration Institute. hdl.handle.net/2019/404

Romme, W.H., C.D. Allen, J.D. Bailey, W.L. Baker, B.T. Bestelmeyer, P.M. Brown, K.S. Eisenhart et al. 2008. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western U.S. Fort Collins: Colorado Forest Restoration Institute. www.cfri.colostate.edu/docs/PJSynthesis.pdf

USDA Forest Service. 2008. FSM 2000–National forest resource management, Chapter 2020–Ecological restoration and resilience. Forest Service Manual Interim Directive No. 2020-2008-1. www.fs.fed.us/im/directives/fsm/2000/id_2020-2008-1.doc

Modeling Long-Term Effects of Altered Fire Regimes following Southern Pine Beetle Outbreaks (North Carolina)

Weimin Xi (Knowledge Engineering Lab, Texas A&M University, College Station, TX 77843, xi@tamu.edu), John D. Waldron (Dept of Environmental Studies, University of West Florida, Ft. Walton Beach, FL 32547), Charles W. Lafon (Dept of Geography, Texas A&M University), David M. Cairns (Dept of Geography), Andrew G. Birt (Knowledge Engineering Lab), Maria D. Tchakerian (Knowledge Engineering Lab), Robert N. Coulson (Knowledge Engineering Lab) and Kier D. Klepzig (U.S. Forest Service Southern Research Station, 2500 Shreveport Hwy, Pineville, LA 71360)

Periodic fires are an important factor shaping the species-rich southern Appalachian forest landscape, and fire regimes in this region have changed significantly over time. The role of fire in maintaining Appalachian forests has been debated and increasingly studied (Delcourt and Delcourt 1998). Experimental studies have shown that pine regeneration increases following prescribed fire (e.g., Vose et al. 1997), and researchers have suggested that reintroducing fire may help to maintain the decreasing natural pine forests (Lafon et al. 2007).

In addition to fire, southern pine beetle (*Dendroctonus frontalis*, SPB) is a major biological disturbance agent affecting pines in this region. For example, from 1999 to 2003, over 400,000 ha (timber value > $1.5 billion) of pine forests were damaged in the southern Appalachians and adjacent Cumberland Plateau. While prescribed fire is increasingly utilized as a means to restore decadent pine forests, the long-term effects of fire following SPB outbreaks are still unclear.

To investigate the synergistic effects of fire and SPB, we used LANDIS-II, a spatially explicit landscape simulation model of forest succession and disturbance. Specifically, we simulated changes in the abundance of pines under SPB disturbance and two fire scenarios: 1) fire suppression, and 2) historic fire regimes prior to fire suppression. Our goal is to understand the long-term effects of fire regimes in pine forest recovery and to provide insights into the effectiveness of post-SPB restoration strategies for the region.

We used the Grandfather Ranger District (GRD) of the Pisgah National Forest of western North Carolina, USA, as our study area. This mountainous region (ca. 777 km²) consists of diverse environmental conditions and high plant diversity and is characterized by extensive hardwood forests. However, pine-oak and pine forests cover about 14.2% of federally managed lands, predominately on dry slopes and ridges. The natural pine species in this region include shortleaf pine (*Pinus echinata*), pitch pine (*P. rigida*), Table Mountain pine (*P. pungens*), Virginia pine (*P. virginiana*), and white pine (*P. strobus*).

This GRD landscape is divided into 11 ecozones (large areas of similar temperature, moisture, and fertility conditions), including the three pine-oak forest dominated ecozones that we focused on in this study (Table 1): I, shortleaf pine-oak forest, which is found primarily at low elevations on broad, exposed landforms in areas with low growing-season rainfall; II, xeric pine-oak and oak forests, on all upper slopes in areas with low dormant-season rainfall and at lower elevations on broad, gentle slopes and ridges; and III, white pine–oak forest, largely at lower

Table 1. Characteristics of three pine-oak ecozones (following Simon et al. 2005) and parameters for two fire regimes (historical fire scenario vs. current fire suppression scenario) in the Grandfather Ranger District, Pisgah National Forest, North Carolina, USA. Fire regime values are estimated means at initiation of LANDIS-II simulations; FRI is fire return interval.

	Ecozone I	Ecozone II	Ecozone III
Dominant Vegetation	Shortleaf pine-oak forest	Xeric pine-oak forest and oak forest	White pine-oak forest
Indicator species	shortleaf pine (*Pinus echinata*) sourwood (*Oxydendrum arboretum*) scarlet oak (*Quercus coccinea*) southern red oak (*Q. falcata)* post oak (*Q. stellata*)	Table Mountain pine (*P. pungens*) scarlet oak pitch pine (*P. rigida*) chestnut oak (*Q. prinus*) mountain laurel (*Kalmia latifolia*)	white pine (*P. strobus*) scarlet oak sourwood chestnut oak blackgum (*Nyssa sylvatica*)
Historic FRI (yr)	5–7	5–7	15–20
Historic Size (ha)	50	50	30
Current FRI (yr)	50	70	90
Current Size (ha)	5	5	3

elevations in areas with higher growing-season rainfall and also exposed upper slopes (Simon et al. 2005).

LANDIS-II simulates large-scale (> 10^5 ha) landscape dynamics and interactions among ecological processes, including succession, seed dispersal, abiotic disturbances (fire and wind), biological disturbance agents (insect outbreaks), and forest management (harvesting) in a forested landscape over long-term (50–500 years) time scales. The landscape in the model is represented as a two-dimensional grid of equal-sized cells (30 × 30 m in our study), which we divided into a simplified mosaic of four existing major forest types (pine, pine-hardwood, hardwood-pine, and hardwood forest) as a starting point for the simulations.

Succession in the model is based on life history attributes of each species, the composition of different species within a cell, and the composition of species in surrounding cells. We parameterized a pool of the 36 most dominant trees and 3 common shrub species within GRD for this simulation using the double exponential algorithm (Scheller et al. 2005) to model seed dispersal. A key parameter for species in LANDIS is an establishment coefficient, which controls the likelihood that a species will establish in a particular cell. We used a finer scale ecosystem process model (LINKAGES) to calculate the establishment coefficients based on the growth and competitive ability of species during first 10-year simulations. In turn, LINKAGES was parameterized using species-specific life history and environmental factors such as temperature, precipitation, growing season degree-days, soil organic matter, nitrogen, and moisture (Xi et al. 2008).

The Biological Disturbance Agent module in LANDIS-II was parameterized to represent the temporal and spatial pattern of SPB outbreaks in this area (Waldron et al. 2007). As a base scenario, we ran simulations with SPB as the only disturbance. This baseline was compared to two fire management and SPB outbreak scenarios: 1) historic fire regime with a mean fire-return interval of 5–20 years; and 2) current fire suppression regime with a mean fire-return interval of 50–90 years (Table 1). Fire regimes, including fire event sizes, ignition probabilities, and fire spread ages for different ecozones, were parameterized based on the published literature and communications with fire experts. Each simulation was run for 500 years.

Our results (Figure 1) indicate that SPB outbreaks alone (i.e., without fire) lead to the disappearance of all pine species from the landscape. Fire suppression promotes the increase of white pine within the landscape, but leads to the reduction of all other pine species. In contrast, historic fire regime favors the natural restoration of shortleaf pine, Table Mountain pine, and pitch pine and reduces the frequency of white pines in the landscape. Our findings are consistent with the hypothesis that SPB and fire disturbance have historically driven succession of pitch pine and Table Mountain pine forests in a beetle-fire-growth cycle, and wildfires are an integral part of the long disturbance regime that forms and maintains pine woodlands in the southern Appalachians (Williams 1998). They also help explain recent (ca. 50 years) increases in the abundance of white pines, which likely benefit from modern fire suppression policies.

Our studies help forest managers and landowners better understand the effects of multiple disturbances on the composition and structure of forests and the potential problems caused by long-term fire suppression policies. Although SPB damage is largely a natural, uncontrollable phenomenon, we have shown that historical fire and fire suppression lead to very different forest compositions. In particular, our projections suggest that frequent fires may assist regeneration and restoration of pine forests damaged by SPB outbreaks, especially species such as shortleaf

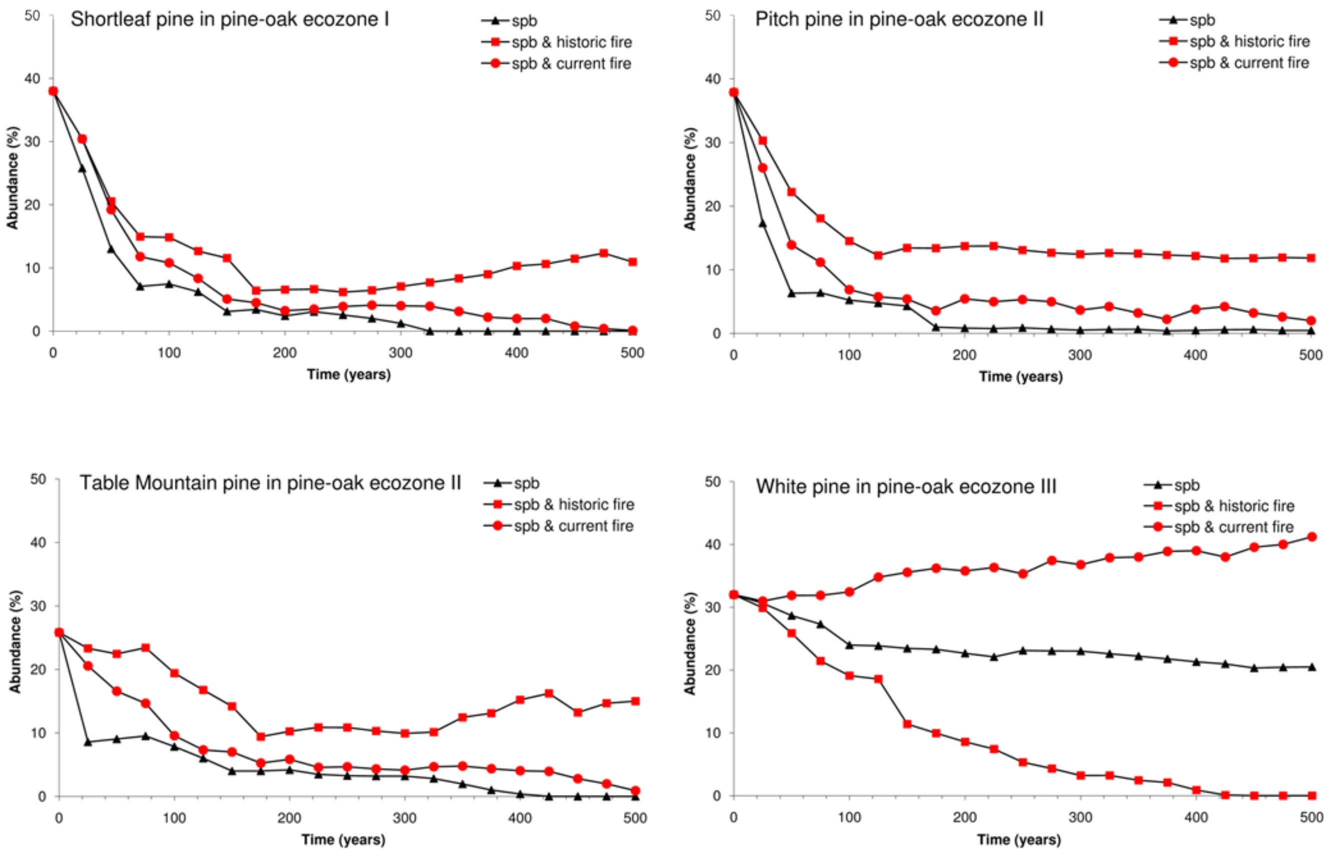

Figure 1. Changes in species abundance (percentage of landscape covered) for four pine species in three pine-oak ecozones over 500 years under two fire regimes in the Grandfather Ranger District, Pisgah National Forest, North Carolina, USA, in a LANDIS-II computer simulation.

pine, Table Mountain pine, and pitch pine, thought to be underrepresented in the present day southern Appalachian landscape. Moreover, fire-based restoration efforts should focus on the shortleaf pine-oak forest, and xeric pine-oak forest, and oak forest ecozones.

Further information about our restoration project is available at http://landscape-restoration.tamu.edu.

Acknowledgments

This research was supported by USDA Forest Service, Southern Research Station cooperative agreement SRS07-CA-11330129-048 and SRS08-CA-11330129-049. We thank Drs. Frost Cecil and Henry McNab for their help on the process of defining fire and ecozone parameters, Drs. Robert Scheller, David Mladenoff, Eric Gustason, and Brian Sturtevant for providing aiding in LANDIS-II use, and Ms. Audrey Bunting and Vickey Chen for their assistance during the project.

References

Delcourt, P.A. and H.R. Delcourt. 1998. The influence of prehistoric human-set fires on oak-chestnut forests in the Southern Appalachians. *Castanea* 63:337–345.

Lafon, C.W., J.D. Waldron, D.M. Cairns, M.D. Tchakerian, R.N. Coulson and K.D. Klepzig. 2007. Modeling the effects of fire on the long-term dynamics and restoration of yellow pine and oak forests in the southern Appalachian Mountains. *Restoration Ecology* 15:400–411.

Scheller, R.M., D.J. Mladenoff, T.R. Crow and T.A. Sickley. 2005. Simulating the effects of fire reintroduction versus continued fire absence on forest composition and landscape structure in the Boundary Waters Canoe Area Wilderness, northern Minnesota, USA. *Ecosystems* 8:396–411.

Simon, S.A., T.K. Collins, G.L. Kauffman, W.H. McNab and C.J. Ulrey. 2005. Ecological zones in the Southern Appalachians: First approximation. USDA Forest Service Research Paper SRS-41.

Vose, J.M., W.T. Swank, B.D. Clinton, R.L. Hendrick and A.E. Major. 1997. Using fire to restore pine-hardwood ecosystems in the southern Appalachians of North Carolina. Pages 149–154 *in* J.A. Greenlee (ed), *Proceedings: Fire Effects on Rare and Endangered Species and Habitats Conference, Couer d'Alene ID, 13–16 November 1995*. Spokane WA: International Association of Wildland Fire.

Waldron, J.D., C.W. Lafon, R.N. Coulson, D.M. Cairns, M.D. Tchakerian, A.G. Birt and K.D. Klepzig. 2007. Simulating the impacts of southern pine beetle and fire on pine dynamics on xeric southern Appalachian landscapes. *Applied Vegetation Science* 10:53–64.

Williams, C.E. 1998. History and status of Table Mountain Pine–Pitch Pine forests of the southern Appalachian Mountains (USA). *Natural Areas Journal* 18:81–90.

Xi, W., R.N. Coulson, J.D. Waldron, M.D. Tchakerian, C.W. Lafon, D.M. Cairns, A.G. Birt and K.D. Klepzig. 2008. Landscape modeling for forest restoration planning and assessment: Lessons from the southern Appalachian Mountains. *Journal of Forestry* 106:191–197.